Journal of the Fisheries Research Board of Canada

Journal de l'office des recherches sur les pêcheries du Canada

Special Issue

Volume 33, No. 3, March 1976

Numéro spécial

Volume 33, nº 3, mars 1976

Lake Erie in the Early Seventies

Environment Canada
Fisheries and Marine Service

Environnement Canada
Service des pêches et des sciences de la mer

Journal of the Fisheries Research Board of Canada

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Environment Canada
Fisheries and Marine Service
Office of the Editor

Environnement Canada
Service des pêches et des sciences de la mer
Bureau du Rédacteur

116 Lisgar, Ottawa K1A 0H3

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Journal of the Fisheries Research Board of Canada

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Journal de l'office des recherches sur les pêcheries du Canada

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Editor's Foreword

In Volume 31(6), June 1974 I outlined the rationale for expanding the scope of the *Journal*. Three new sections have since been successfully introduced — *Perspectives*: Informal essays of opinion or hypothesis that are of broad Canadian or international interest on fisheries management, ocean science, or the aquatic environment; *Letters to the Editor*: Comments on published articles; and *Book Reviews*. I also stated that we would consider further expansion of the *Journal* in the future.

This seems a desirable time to reaffirm our broad scope. Although our editorial policy statement and guide for authors clearly outline in each issue that our scope includes ocean sciences and the aquatic environment as well as fisheries, scientists in the physical and chemical disciplines in particular, apparently remain unaware of our broad scope, and this is reflected in the overwhelming biological content of the *Journal*. In the 1950s the *Journal* published most of the results of Canadian limnological and oceanographic research because most limnologists and oceanographers were employed by the Fisheries Research Board, but there was an understandable drift to specialist journals in the 1960s. However, the *Journal* still publishes more papers on physical oceanography and limnology than any other Canadian journal.

Recent government reorganization has brought us full circle from the turn of the century when the Department of Marine and Fisheries was in existence. Within the Fisheries and Marine Service we are now responsible for scientific and technical information in both Fisheries Management and Ocean and Aquatic Sciences. This broad mandate needs to be reflected in the content of the *Journal*. Scientists should note, therefore, that articles making original contributions to any aspect of marine and freshwater science covering relevant work in biology (especially fisheries science), chemistry, geology, geophysics, limnology, or physical oceanography, that are relevant to Canada, will be considered for publication.

In keeping with the intent of the foregoing statements it is my pleasure to introduce this Special issue on studies conducted in Lake Erie in 1970 and 1971. The Lake Erie group represents an interdisciplinary team from various organizations in Ontario and what began as surveys primarily intended to produce baseline limnological data in the disciplines of physics, geology,

Printed in Canada (J4063)
Imprimé au Canada (J4063)

chemistry, and biology, ended up as a coordinated study. Hopefully, this special issue will help increase the understanding of the whole-lake processes.

J. C. STEVENSON,
Editor/Director of
Scientific Information

Note du directeur

Dans le volume 31(6) de juin 1974, j'ai expliqué pourquoi nous élargissions la portée du *Journal*. Depuis lors, nous y avons incorporé avec succès trois nouvelles sections: *Essais:* opinions et hypothèses d'un intérêt général pour le Canada et l'extérieur sur la gestion des pêches, les sciences de la mer et le milieu aquatique; *Lettres à la rédaction*: commentaires sur les articles publiés; et *Critiques*. J'ai en outre précisé que nous étions disposés à prendre en considération une plus grande expansion du Journal à l'avenir.

Il serait maintenant bon d'insister à nouveau sur l'envergure de notre revue. Même si l'exposé de notre politique de publication et notre guide des auteurs précisent, dans chaque numéro que nous nous intéressons aussi bien aux sciences de la mer et au milieu aquatique qu'aux pêches, les scientifiques du monde de la physique et de la chimie ne semblent pas au courant de notre expansion: c'est ainsi que les articles de biologie restent beaucoup plus nombreux que les autres. Durant les années 50, le *Journal* publiait presque tous les résultats des recherches limnologiques et océanographiques parce que l'Office des recherches sur les pêcheries employait la plupart des limnologistes et des océanographes, mais il y eu un mouvement progressif vers les publications spécialisées au cours des années 60. Cependant, le *Journal* publie toujours plus d'articles sur l'océanographie et la limnologie physique que toute autre revue canadienne.

La récente réorganisation gouvernementale nous a réattribué les tâches que nous avions au début du siècle, lorsque le ministère de la Mer et des Pêcheries existait. En effet, le service des pêches et de la mer s'occupe maintenant de l'information scientifique et technique pour la Gestion des pêches ainsi que les Sciences océaniques et aquatiques. Il est donc normal que le *Journal* reflète cet important mandat. Les scientifiques doivent par conséquent noter que nous prendrons en considération tout article inédit traitant d'un quelconque aspect des sciences de la mer et des eaux douces et faisant état de travaux de biologie (surtout sur la pêche), de chimie, de géologie, de géophysique, de limnologie ou d'océanographie physique qui concernent le Canada.

Pour continuer dans la même ligne de pensée, c'est pour moi un plaisir que de présenter cette publication spéciale sur des recherches menées au lac Érié en 1970 et 1971. Le groupe d'étude du lac Érié consiste en une équipe interdisciplinaire de divers organismes d'Ontario ce qui a abouti à un travail bien coordonné était au départ une série de relevés destinés à procurer les données limnologiques de base dans des disciplines telles que la physique, la géologie, la chimie et la biologie. Nous espérons que ce numéro spécial aidera à augmenter la compréhension de l'ensemble des phénomènes lacustres.

J. C. STEVENSON
Le rédacteur et directeur
de l'information scientifique

Preface

The need for more information in respect to the water quality of Lake Erie was indicated in the reports written by USFWPCA (1968a and 1968b), and by the Report of the International Lake Erie Water Pollution Board (1969). In addition, although many investigations of limnological processes in Lake Erie have been carried out (Herdendorf et al. 1974), lakewide survey data has remained sparse. In response to the concerns about the deteriorating water quality of Lake Erie, the staff of the Canada Centre for Inland Waters, Burlington, Ontario (C.C.I.W.), planned detailed studies of the lake for 1970 and 1971. It is hoped that this publication, resulting from these studies, will prove to be a useful advance in our knowledge of the lake.

Seven research cruises covering the Central basin were implemented to investigate the oxygen depletion processes during August 1970. Ten lakewide survey cruises during the 9 month ice-free season from April to December 1970 provided data on the water chemistry and the phytoplankton and zooplankton distributions within the lake. An extensive survey of the sediments of Lake Erie, four more lakewide chemical and biological surveys, and seven brief geochemical research cruises were conducted in 1971. A number of publications of a descriptive nature have resulted from these surveys and are referred to in the other papers appearing in this issue.

The 1970–71 Lake Erie surveys were primarily intended to produce baseline limnological data in each of the four traditional disciplines of physics, geology, chemistry, and biology and were not planned so that the results would link together to form a large interdisciplinary study. After the initial limnological descriptions were complete, the scientists continued to work on their data in an attempt to describe lake processes in some depth. However, many found difficulty in doing so while they remained within the confines of their own data sets and this led many of the workers into discussions with their colleagues at C.C.I.W.

These circumstances led to the formation of the Lake Erie Group, consisting mostly of scientists from C.C.I.W. and the Lake Erie Fisheries Research Station of the Ontario Ministry of the Natural Resources. The Group members hoped that they might begin to quantify some of the processes requiring input from scientists of different disciplines.

Once a week, the Lake Erie Group met to discuss material which had been prepared by a member and distributed to the Group prior to the meeting. Subsequent rewriting then took into account the many comments and ideas voiced during the discussions with the final result being the papers which form this special issue.

A major benefit of the Group discussions was the increased understanding and knowledge of whole-lake processes gained by the individual Group members. However, the extra effort and time involved in such interdisciplinary work should not be underestimated. The elucidation of the interrelationships between the different areas of study has proven more difficult than anticipated and progress has been limited. The results which have been achieved are reported in the final paper of this issue. A major problem in this work arises from the difficulty of communication between scientists working in different disciplines. The difficulty arises partly out of a lack of a common technical language and background of reference, but more particularly from the different thought patterns which develop in persons of different academic background. Interdisciplinary work requires patience and personal adaptation. Also, the full investigation of a lake process soon reveals a number of unexpected complexities in the process and real progress must inevitably be slower than anticipated.

The guidelines for the papers of this issue are a little different from those customarily followed in the primary scientific literature. First, there is much new information upon

Printed in Canada (J3990)
Imprimé au Canada (J3990)

which our discussions are based and, for this reason, more data have been included in some of the sections than would normally be found in the primary literature. Second, readers interested in Lake Erie as a whole will perhaps read some of the sections which are removed from their normal scientific interest. In this regard, the use of specific terms which are not well known has been kept to a minimum and an effort made to make the papers more readable to someone exploring topics outside of his immediate study area. Unfortunately, in achieving this requirement, some brevity has been lost and each paper has been summarized under the heading of "Summary and Conclusions" to compensate for this.

HERDENDORF, C. C., S. M. HARTLEY, AND L. J. CHARLESWORTH. 1974. Lake Erie bibliography in environmental sciences. Bull. Ohio Bio. Soc. 4: 1–113.

INTERNATIONAL LAKE ERIE WATER POLLUTION BOARD. 1969. Report to the International Joint Commission on the Pollution of Lake Erie, Lake Ontario and the International Section of the St. Lawrence River, Vol. 2, Lake Erie, 277 p.

UNITED STATES FEDERAL WATER POLLUTION CONTROL AGENCY. 1968a. Lake Erie environmental summary 1963–1964, 160 p.

1968b. Lake Erie report: A plan for water pollution control, 86 p.

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Préface

Les rapports de l'USWPCA (1968a et 1968b) et le *Report of the International Lake Erie Water Pollution Board* (1969) font ressortir le besoin d'en connaître davantage sur la qualité de l'eau du lac Erié. De plus, bien qu'il y ait eu des études de faites sur les processus limnologiques du lac Erié (Herdendorf et al. 1974), les relevés couvrant tout le lac sont encore rares. L'inquiétude causée par la détérioration de la qualité de l'eau du lac Erié a incité le personnel du Centre canadien des eaux intérieures, Burlington, Ont. (CCEI) à entreprendre des études détaillées du lac en 1970 et 1971. La présente publication est le fruit de ces études. Nous espérons qu'elle contribuera à augmenter nos connaissances du lac.

Sept croisières ont été effectuées dans le bassin central en vue d'étudier les processus d'épuisement d'O_2 en août 1970. Dix croisières à la grandeur du lac, réparties sur les 9 mois au cours desquels le lac est libre de glace, soit avril à décembre 1970, ont permis de cueillir des données sur la chimie de l'eau et la répartition du phytoplancton et du zooplancton. En 1971, on a fait un relevé détaillé des sédiments du lac Erié, quatre autres reconnaissances chimiques et biologiques, et sept courtes croisières de recherche géochimique. Ces relevés ont été l'objet de plusieurs publications de nature descriptive, et on y réfère dans les articles qui apparaissent dans ce numéro.

Les relevés de 1970–71 avaient comme objectif principal de produire des données limnologiques de base dans chacune des quatre disciplines traditionnelles, physique, géologie, chimie et biologie. Le travail n'a pas été planifié de façon à fusionner les résultats et à produire une grande étude interdisciplinaire. Après avoir terminé les descriptions limnologiques initiales, les scientifiques ont poursuivi l'analyse de leurs données afin de décrire assez en détail les processus du lac. Cependant, plusieurs ont eu des difficultés aussi longtemps qu'ils se sont confinés à leurs propres données. Ceci a incité plusieurs chercheurs à discuter de leurs problèmes avec leurs collègues du CCEI.

Ces circonstances ont abouti à la formation du Groupe du lac Erié, comprenant en grande partie des scientifiques de CCEI et de la Station de recherches sur les pêches du ministère des Ressources naturelles de l'Ontario. Les membres du Groupe espéraient de cette façon pouvoir quantifier certains processus nécessitant la contribution de scientifiques de différentes disciplines.

Le Groupe du lac Erié se réunit une fois par semaine pour discuter du matériel préparé par un des membres et distribué aux autres membres avant la réunion. On a ensuite corrigé et rédigé de nouveau le travail pour incorporer les nombreux commentaires et idées qui jaillirent de ces discussions et préparé une version finale, celle qui apparaît dans la présente publication.

Le bénéfice majeur retiré de ces discussions a été de permettre aux membres de mieux comprendre les processus du lac tout entier. Il ne faudrait toutefois pas sous-estimer l'effort et le temps additionnels qu'implique un travail interdisciplinaire de ce genre. L'élucidation des interactions entre les différents domaines d'étude s'est avérée plus difficile que prévue, et les progrès ont été limités. Le dernier article de ce numéro donne un compte rendu des réalisations. Un problème majeur dans un travail comme celui-ci provient de la difficulté qu'ont les scientifiques de disciplines différentes de communiquer entre eux. Ceci est dû en partie à un manque de références de base et de vocabulaire technique communs aux différentes disciplines, mais surtout à une orientation différente de la pensée qui se développe chez des personnes de formation académique différente. Le travail interdisciplinaire exige de la patience et du tact. De plus, l'étude complète des processus d'un lac ne tarde pas à révéler nombre de complications inattendues, et le progrès réel est beaucoup plus lent que prévu.

Les normes utilisées dans la préparation des articles qui constituent cette publication spéciale diffèrent quelque peu de celles ordinairement suivies dans les publications

scientifiques de fonds. En premier lieu, les discussions sont fondées sur beaucoup d'information inédite. Pour cette raison, certaines sections contiennent plus de données qu'on ne trouve normalement dans des publications de fonds. En second lieu, les lecteurs intéressés au lac Erié dans son ensemble liront peut-être quelques-unes des sections débordant leur sphère d'intérêt scientifique normal. Sous ce rapport, les termes specifiques pas très bien connus ont été réduits au minimum, et on s'est efforcé de rendre les articles aussi faciles à lire que possible pour les personnes qui explorent des sujets en dehors de leur domaine d'étude immédiat. Malheureusement, le travail en a souffert sous le rapport de la brièveté. Pour compenser, chaque article a été résumé sous la rubrique « Summary and Conclusions ».

HERDENDORF, C. C., S. M. HARTLEY, AND L. J. CHARLESWORTH. 1974. Lake Erie bibliography in environmental sciences. Bull. Ohio Bio. Soc. 4: 1–113.

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Lake Erie and its Basin

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SLY, P. G. 1976. Lake Erie and its basin. J. Fish. Res. Board Can. 33: 355–370.

The bathymetry of Lake Erie is separated into Western, Central, and Eastern basins, whose form and feature have been strongly influenced by bedrock geology and the effects of Pleistocene events. The climatic regime is mid-continental but is modified by the overall effect of the Great Lakes, mean annual temperatures have a range of about 21 C. European settlement in the basin began in the early 1800s and land use patterns and historic settlement differ distinctly north and south of the lake. Changes in Lake Erie water quality are strongly associated with both agricultural and industrial development, but did not become severe until towards the end of the 1930s. However, changes and depletion of fish stocks likely owe as much to overfishing as to changes in water quality.

SLY, P. G. 1976. Lake Erie and its basin. J. Fish. Res. Board Can. 33: 355–370.

Au point de vue bathymétrique, le lac Erié se subdivise en trois bassins, ouest, central et est, dont les formes et caractéristiques ont été fortement influencées par la géologie du socle rocheux et les événements du Pléistocène. Le régime climatique est mi-continental, mais il est modifié sous l'influence générale des Grands lacs. Les températures annuelles moyennes ont un écart d'environ 21 C.

La colonisation du bassin par les Européens débuta tôt en 1800, et l'utilisation des terres aussi bien que l'évolution de cette colonisation sont nettement distinctes au nord et au sud du lac. Les changements dans la qualité de l'eau du lac Erié sont étroitement liés au développement tant agricole qu'industriel, mais ne devinrent sérieux que vers la fin des années 1930. Par ailleurs, les changements et l'épuisement des stocks de poissons sont vraisemblablement le résultat d'une surexploitation tout autant que d'une détérioration de la qualité de l'eau.

Accepted November 17, 1975

Accepté le 17 novembre 1975

LAKE Erie is the southernmost lake in the Laurentian Great Lakes System, it has an area of 25,300 km^2, a volume of 470 km^3, and its surface is 174 m a.s.l. (above sea level). Much of the lake is shallow (Fig. 1); the west part of the lake is mostly less than 11 m in depth, the central part is little more than 25 m in depth, only in the east part is there a significant depth of water where maximum soundings reach 64 m. The lake has a general orientation of west-southwest, east-northeast, and has a maximum length of more than 390 km; the width averages about 70 km but reaches a mid-lake maximum of about 90 km. Although Lake Erie lies south of latitude 43°N its location in mid-continent exposes it to a considerable range of climatic conditions. In midwinter, mean daily temperatures average −3 to −2 C over the lake and extensive ice cover frequently develops. In summer the mean daily temperature averages between +21 and +23 C. Winds from south and southwest quadrants predominate throughout the year.

Land use patterns in the watershed surrounding the lake show that along the north shoreline, and inland areas, agricultural practice is intensive, and that there are relatively few centers of large population or intensive industrial activity. Along the south shoreline the Lake Erie watershed is very narrow (Fig. 2), except at the west end of the lake. In this area agricultural land use is also intensive but there are a number of large population centers with related intense, and heavy, industrial development.

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Imprimé au Canada (J3991)

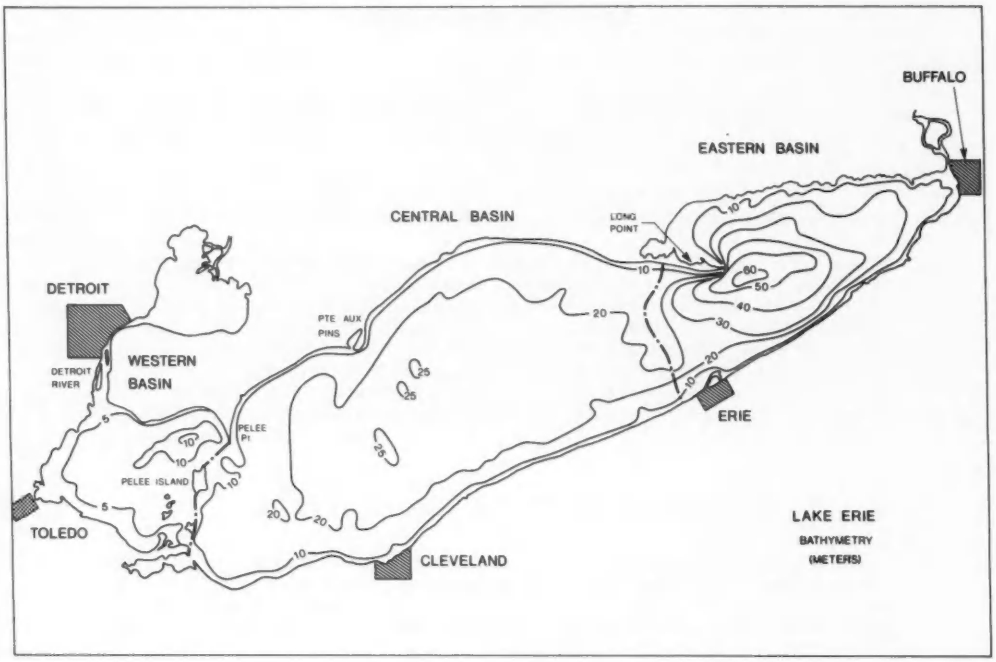


FIG. 1. Lake Erie bathymetry.

European settlement of the Lake Erie watershed began in the early 1800s on both sides of the international boundary, and it may be assumed that localized degradation of the lake waters was apparent even before the 1850s. Lake water quality has shown a progressive degradation since that time but conditions may be considered to have been generally acceptable until some period towards the end of the 1930s. Lake Erie water quality continued to decrease, likely at an increasing rate related to population increase, land use activities, and use of detergents, and has culminated in the highly eutrophic state and associated extensive anoxic conditions observed during lake studies in the early 1970s. Recognition of this situation prompted the signing of the Canada/U.S. Water Quality Agreement in 1972, and the initiation of massive remedial action.

Geological Development of the Lake Erie Basin

The Lake Erie basin is underlain by a considerable thickness of Palaeozoic sediments (Fig. 3). Dolomitic limestones and shales of Silurian age form the Niagara Peninsula and limit the basin in the east. Upper and Lower Devonian formations underlie much of the north part of the basin and in the south, Lower Devonian limestones give way to upper Devonian shales for much of the length of the basin. In the most southerly part of the region, sandstones and shales of lower Carboniferous age occur.

Structurally (Fig. 4) the region is influenced by three major elements (Hough 1958). To the south, the lithological and structural control of the Appalachian geosyncline influences the general orientation of the existing lake basin, whilst in the west, the basin is linked to another structure by the presence of the Findlay Arch. This Arch, which up-warps a rim of Lower Devonian dolomitic rocks, largely controls the form of western Lake Erie. The chain of islands and the underwater rise which separates the Central and Western basins of the lake, reflect this. No evidence now exists in the basin of sedimentary formations which date between early Carboniferous and Pleistocene age. By implication, however, it is believed that the region was largely subject to subaerial weathering during Tertiary and early

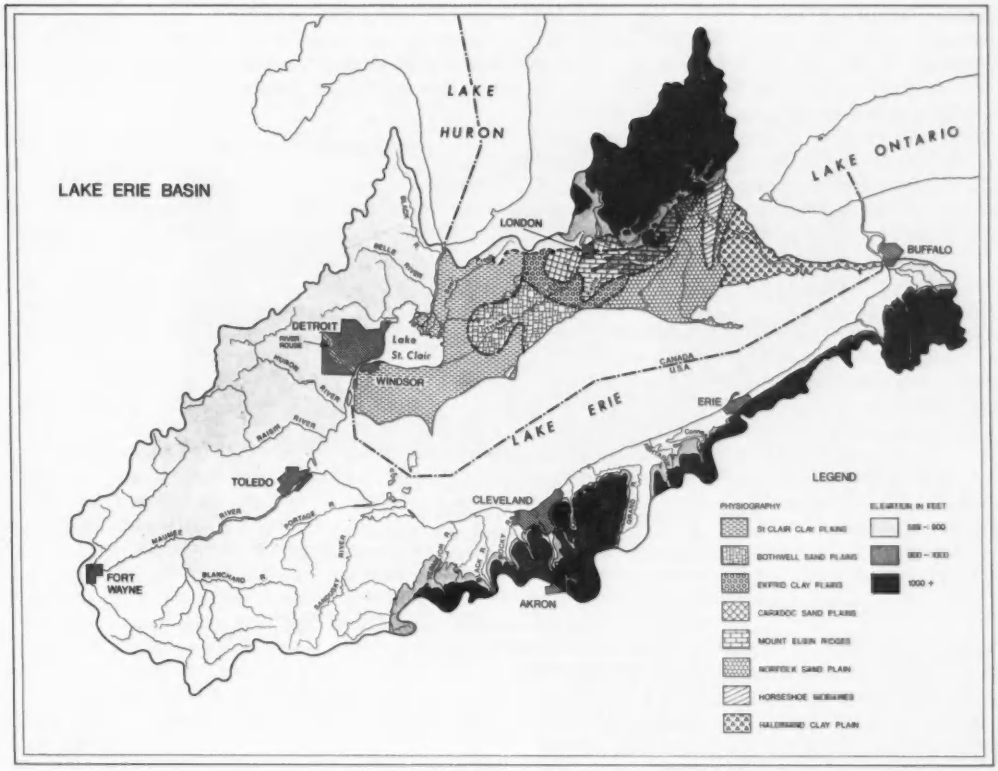


FIG. 2. Lake Erie Basin — physiography and relief.

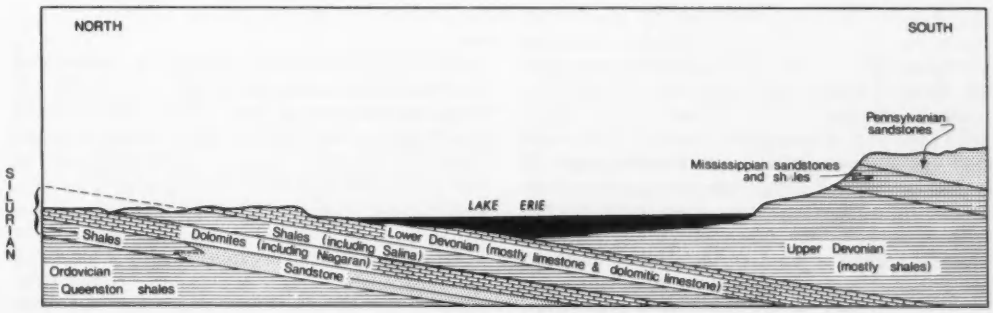
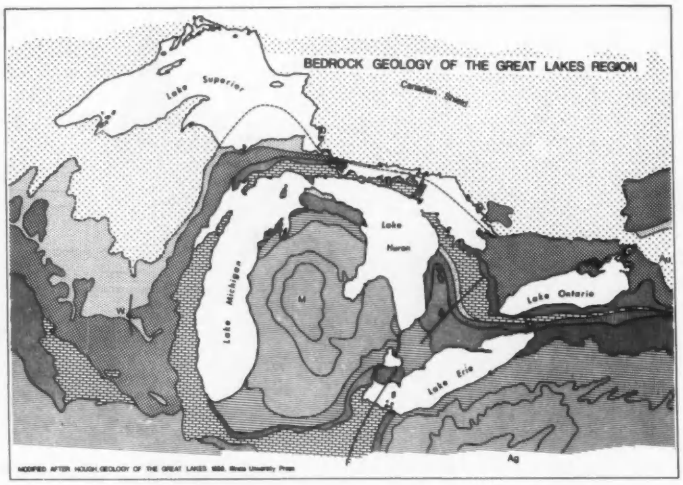


FIG. 3. Bedrock stratigraphy cross section Lake Erie basin. (Modified from Hough 1958.)

Quaternary times and there is tentative evidence (Hough 1958) to suggest that the evolution of the Lake Erie basin, as we know it today, may have begun in the Tertiary period. Reconstruction of early drainage patterns in the Great Lakes region suggests that one tributary of the Proto-St. Lawrence aligned itself along the axis of present Lake Erie, perhaps including the upper part of the Maumee River, and crossed the Niagara Peninsula at its western extremity.


BEDROCK GEOLOGY OF THE GREAT LAKES REGION
Lake Superior
Canadian Shield
Lake Michigan
Lake Huron
Lake Ontario
Lake Erie
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M MICHIGAN BASIN
W WISCONSIN ARCH

FIG. 4. Bedrock geology of the Great Lakes Region. (Modified from Hough 1958.)

"Excavation" of the Lake Erie basin began with the onset of Quaternary glaciation, under severe conditions of glacial scour and frost shatter. There have been four major glacial events in the region (Sly and Thomas 1974), and, between each, extended periods of temperate climatic conditions similar to, or warmer than, those now existing. In addition to the movement of ice and the interactive processes of climatic weathering the basin has been subjected to differential rock stress as a result of ice loading and subsequent isostatic rebound. The result of these conditions has been to lower the bedrock surface of the east and central parts of the basin relative to that of the west. The existing lake bed reflects this condition but the existence of considerable infill by glacial and postglacial materials lessens the effect. The thickness of glacial infill in the Western basin is generally less than 13 m, in the Central basin 30 m, and in parts of the Eastern basin there is more than 40 m of material. Equivalent basin maximum thickness of the overlying postglacial sediment are 8, 20, and 26 m, respectively (Hobson et al. 1969; Sly and Lewis 1972; and C. F. M. Lewis personal communication).

RESOURCE MATERIALS

The Palaeozoic formations of the Erie basin remain the source of a number of economically important resources. The cement and aggregate industries utilize many exposures of limestone materials, and the building industry uses selective outcrops of both limestones and sandstones. Additionally, exploitation is focussed on the salt deposits (salina formation) of upper Silurian age which underlie much of the region. Associated with these Palaeozoics, also, are limited reserves of hydrocarbons. Most of the gas reserves in the basin have already been tapped, although some

drilling continues in offshore locations. Oil was discovered at Petrolia (near Sarnia) in 1858, and production began a year later; production within the Erie basin today, however, is minimal.

Pleistocene Events and Physiographic Evolution

The development of the detailed physiographic features in the Erie basin has been greatly affected by the immediate postglacial history of the Great Lakes and, in particular, by successive high and low lake levels, which formed as a result of ice impoundment and outlet control (Sly and Thomas 1974).

The sequence of known lake stages in the Erie basin began with retreat of the Erie ice lobe from the Fort Wayne moraine in Michigan, Indiana, and Ohio about 14,000 yr ago. Meltwater, ponded between the ice and this moraine, formed Lake Maumee I at 244 m a.s.l. which discharged westward to the Mississippi system. General retreat with minor re-advance during the next 500–1000 yr gradually enlarged the area inundated by lake waters, until the whole basin was probably submerged. It is likely that the Pelee and Erieau moraines (Fig. 5) and possibly the Port Maitland moraine were constructed at this time. Subsequently, lake levels dropped and fluctuated as new outlets were opened and old outlets were deepened.

About 13,000 yr ago, ice re-advanced into the Eire depression. This ice re-excavated the east basin, constructed the Norfolk moraine, and raised the water level in the remainder of the basin to the Lake Whittlesey stage at 225 m a.s.l. A period of rapid retreat followed in which ice gradually evacuated the Erie basin and parts of the adjacent Michigan, Huron, and Ontario basins. New outlets through the Huron and Michigan basins, opened by this retreat, allowed lake levels to fall to Early Algonquin Lake stages.

By 12,600 years BP (before present), ice had retreated sufficiently to free the Mohawk–Hudson Valley route and allow Lake Iroquois to come into existence in the Ontario basin. Simultaneously, Erie water levels fell as they drained over the Niagara Escarpment into Lake Iroquois via the modern Niagara River. This event marked the end of glacial influence in the Erie basin and brought into being Early Lake Erie, a lake whose altitude was, and is, controlled by the elevation of the sill at Buffalo.

Continued glacial retreat in central Ontario indirectly affected phases of Early Lake Erie by alternating the flow of water through it. With ice retreat from the Trent Valley and North Bay outlets, the upper Great Lakes discharge bypassed Lake Erie. This reduced Erie's inflow to stream discharge from within its own small watershed and probably reduced its outflow by 90% (Sly and Lewis 1972). Upper Great Lakes discharge was progressively restored to Lake Erie by the St. Clair–Detroit River route during the Nipissing Great Lakes stage, beginning about 6000 yrs ago.

Part of the Eastern basin around Buffalo was still depressed 35 m or more, relative to Central and Western basins, due to differential recovery from ice loading when Early Lake Erie came into existence. Because the water level in this lake stood at about 135 m a.s.l., the Western and Central basins were effectively drained. Water ponded in the deepest parts of the Central basin behind the Pelee and Norfolk moraines and discharged easterly over them, incising channels across their southern termini. The outlet at Buffalo rose rapidly; by 11,000 yr BP, the water level had risen from about 38 m to about 18 m below present lake level. Water levels subsequently rose more slowly except for a rapid, 8 m jump about 4000 yr ago when Upper Great Lakes discharge was fully restored through Lake Erie (174 m a.s.l.).

Early Lake Erie, with its drastically reduced volume and inflow, must have warmed rapidly and likely became very productive. Waves of the low-level lake attacked the older tills and proglacial lake deposits, and winnowed, transported, and redeposited the silt and clay in deeper areas. The residual sands were carried by littoral drift along the low shoreline and deposited as beaches and spits. Tributary streams deepened their valleys toward the lower base level and poured their sediments into the lake. Some sand and gravel deposits of this lake stage are still exposed in the lake bed today; others are buried beneath the mud deposits of more recent phases of Lake Erie; still others such as the modern spits — Pelee, Point aux Pins, Long Point, Presqu' Isle — have maintained themselves and shifted laterally as they kept pace with the rising waters of the lake. Rising base levels in the Central and Western basins caused the lake to inundate low-level marshy shore zones and to flood the tributary streams and infill their lower valleys with fine-grained sediments. Wave activity associated with the rising levels of Lake Erie eroded the slopes of the basin at successively higher levels, planed off topographic irregularities, produced a gently sloping sand-littered terrace, and carved shore bluffs up to 38 m high in the unconsolidated deposits surrounding the basin. Today, fine-grained sediments have filled the offshore areas of Western and Central Lake Erie to a near level surface, completely

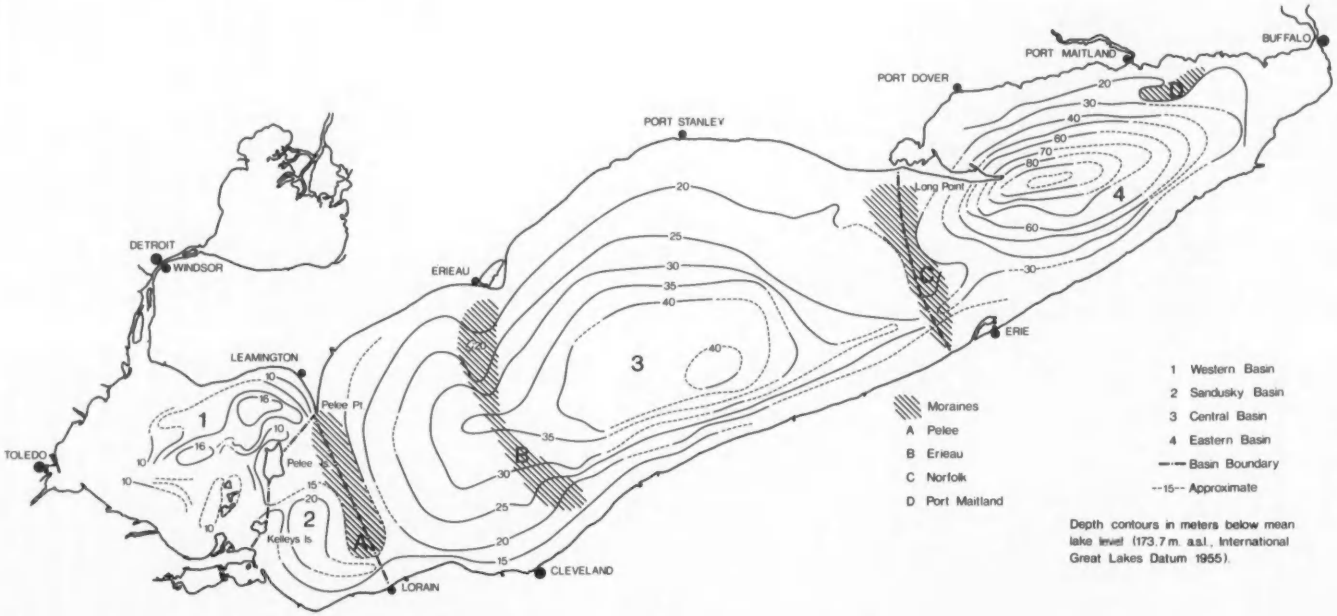

FIG. 5. Inferred bathymetry of Early Lake Erie showing moraines. Contours drawn on top of glacial deposits relative to present lake level. (After Sly and Lewis 1972.)

obliterating topographic details of the old lake bed.

The character of the lake basin remains extremely responsive to the modifying processes of sediment erosion-transportation and deposition, and it is estimated that the average maximum rate of mud accumulation, during the last 12,000 yr, has been about 0.03, 0.19, and 0.3 cm/yr (Sly and Lewis 1972), respectively, in the Western, Central and Eastern basins of the lake. (It should be noted, however, that these average rates include a period of time when much of the basin was subaerially exposed. Rates of deposition today, under subaqueous conditions, are therefore much higher than such averaged, long-term values.) If the present conditions are considered it may be seen that, with an estimated sediment accumulation rate of about 0.2–0.3 cm/yr in the Western basin of Lake Erie (A. L. W. Kemp personal communication) and shoreline erosion rates of up to 1.5 m/yr (Great Lakes Levels Board 1968; surveys and unpublished data; W. S. Haras personal communication) significant changes could result during a water level stability period of 2000 yr or more. An uncorrected first-order estimate based upon such values could imply a shallowing of the basin by 30–35% and an increase in its surface area by as much as 15–20%.

In eastern Lake Erie, Long Point continues to extend southeastwards at a rate of about 7 m/yr (Sly and Lewis 1972) and thereby adds another 6–8% to the area of the existing bay on the north side, in the form of a protected zone, each 100 yr. On the south shore of Lake Erie, at Presqu'Isle peninsula, a corresponding eastward migration of about 5.5 m/yr was recorded by Berg and Duane (1968).

Characteristics of the Watershed

The Lake Erie watershed is extremely narrow over much of the length of the lake (if one excludes the Lake St. Clair and Thames River drainage basin, but expands in the west to incorporate the drainage of western Ohio, and neighboring Michigan and Indiana). The Maumee River system, whose sediment load is of major importance, dominates this part of the basin, with lesser contributions from the Huron, Raisin, Portage, and Sandusky rivers. These rivers drain the flat but gently rising old lake deposits which provide the basis for intensive agricultural practice in the area; in this respect it should also be noted that man-made drainage of swamp and marshlands on the Maumee system has almost certainly led to even greater sediment loading downstream (Monke and Beasley 1975).

East of this area the highlands of northern Ohio, Pennsylvania, and western New York force the watershed divide to approach the lake to within 6 km at a point east of Erie, Pa. The larger rivers, here, include the Cuyahoga, the Grand (Ohio), and Cattaraugus Creek. The land rises strongly from the lakeshore and is generally traversed by numerous short and relatively steep valleys. Only at the eastern extremity of the lake, where the gentle dip slope of the Niagara Escarpment begins to affect the regional morphology, do flat lands again predominate.

Generally speaking, the watershed to the north of Lake Erie shows considerably less relief than that which lies to the south and with the exception of the Grand River (Ont.), at the northeastern end of the basin, there are no rivers of any size which enter the lake; although in the central part of the north shore the bluff shoreline is traversed by a number of steeply cut valleys, the most notable of which is Kettle Creek which enters the lake at Port Stanley.

The northern part of the Erie watershed has been examined in detail by Chapman and Putnam (1966), and may be used as an example of the complexity of landform which surrounds the basin (Fig. 2).

In the western part of the Ontario watershed the St. Clair clay plains cover an area of about 5800 km^2. The region is generally flat, although the Blenheim and Charing Cross moraines locally develop 15–30 m of relief. Most soils are derived from shallow deposits of lacustrine silty clay which overlie a levelled surface of glacial till. Drainage is poorly developed and dredged ditches and tile under-drains are essential to maintain agricultural use, providing a wide variety of cash crops.

East of this area the Bothwell sand plain occurs in the form of thinly layered sands which overlie clay. These sands originated as deltaic sediments of the Thames River during the period of glacial Lake Warren. The region covers some 1800 km^2 and in patches, where the sand cover is very thin, standing water and swamps overlie the clay. Generally the agricultural production from this soil type is low and much of the wetter land remains wooded.

Eastwards, again, the Ekfrid clay plain covers an area of about 900 km^2. The land is flat and the thin lacustrine clays are often silty and highly calcareous. The soils are generally well textured in the better drained parts and commonly produce dark-surfaced clay loams; much of the land is used for pasture.

The Mount Elgin ridges form the watershed divide between the upper Thames River (near

London) and the Norfolk sand plain and they cover an area of some 1400 km². Locally, relief exceeds 30 m and the land rises generally to the northeast to reach an elevation of more than 230 m above lake level. The ridges are formed by the various moraines (Ingersoll, St. Thomas, Norwich, Tilsonburg, Paris, and Galt) of Lake Wisconsin age. Drainage is often incomplete and in lowlands, swamp, marsh, and peat areas exist. Soils on the slopes of the ridges frequently show signs of erosion and gully effects are common. Upland soils are intensely cultivated; lowland areas are mostly used for pasture.

The Norfolk sand plain covers an area of about 3100 km² and forms much of the land adjacent to Long Point. The sands and silts of this area were deposited as a delta in glacial lakes Whittlesey and Warren and reach maximum thickness of about 25 m over till and morainic materials; the average thickness is about 10 m. Most of the drainage to Lake Erie is by short creeks whose valleys may be incised as much as 30 m into the surrounding plain. To the north, drainage runs into the Grand River. Although drainage in the plain is incomplete, well water is abundant and of good quality.

Early agricultural development in the Norfolk plain rapidly exhausted the lighter soils and erosion problems became extensive. Farm abandonment was a severe problem in this area. However, land reclamation, forestation, and improved agricultural practices have turned this region into one of high productivity and the cultivation of tobacco, cereal crops, fruits, and vegetables are now of major importance.

The Haldimand clay plain lies between the Niagara Escarpment and Lake Erie, at the eastern end of the lake, and extends over some 3400 km². Most of the soil is composed of a partial mixture of thin lacustrine clay, and underlying clay till. Locally, slight relief is provided by swells in the ground moraine. The plain dips gently southwards. The composition of the underlying bedrock is frequently reflected in the composition of surface materials. Drainage is generally inadequate and wetlands, and peat and bog patches, are common. The soils are variable depending much upon the character of the ground moraine, and sandy and gravelly areas give rise to lighter soils which offer an excellent base for fruit and vegetable production. Elsewhere pasture and cereal production predominate.

The Grand River drains an area of some 6600 km² on the dip slope of the Niagara Escarpment and intersects both the Haldimand and Norfolk plains, it has a length of about 290 km. The upper reaches drain a complex of glacial spillways formed in the till plains of central, southern Ontario. In the extreme upper reaches there has been little valley cutting, and grading has been controlled by bedrock exposure. The river system has been partially dammed in its upper reaches to form storage reservoirs. In its lower reaches the Grand has formed its own channel as it crosses the lake plains, and below Brantford the gradient averages about 0.4 m/km. Much of the flow of the lower Grand River is also controlled as a prevention against flooding.

Climatic Conditions in the Erie Basin

Climatic conditions in the Erie basin probably reached their warmest postglacial values at some time between 8000 and 4000 yr BP, and likely conditions today compare closely with those which were in effect about 9000 yr ago. The short duration of this climatic maxima should be recognized, since barely 3000 yr previously the region was under the influence of the last glacial advance in the area (Port Huron Stage). Although Beeton (1961) has shown that there has been a general warming of Lake Erie, by about 1 C, since 1918, it seems likely that future climatic conditions will show a slight but general decrease in average annual maxima; however, no immediate or rapid downturn in temperature is projected (Chin and Yevjevich 1974).

Surface air temperatures show significant modification by lake influences all around the periphery of Lake Erie; and in addition there is a discernible north/south temperature gradient which amounts to about 2 C throughout the seasons; a similar east/west gradient also exists (warmer towards the west). Mean daily temperature ranges vary between 9 and 11 C over much of the landward portion of the basin, but remains between 6 and 9 C over most of the lake surface (Phillips and McCulloch 1972).

In winter, mean daily temperatures range between −2 in the southwestern part of the basin and −4 C in the northeastern areas; and in normal winters most of the lake surface is frozen over. Equivalent summer temperatures range between 23 and 21 C.

Precipitation, which runs between 5.0 and 7.5 cm/mo, is evenly spread throughout the year. There is a discernible minima in February but no well defined maxima. There is clearly an increase in precipitation from west to east across the lake, and records for the years 1964–73 (supplied by the Atmospheric Environment Service, Canada) show that average total values in the Western basin of Lake Erie were 82 cm, in the Central basin 89 cm, and in the Eastern basin 92 cm. These values

presumably also reflect the seasonal dominance of westerly winds and increasing downwind precipitation. Trends in precipitation data over the landward components of the basin are less easily defined due to the effects of additional physiographic controls, and the influence from other adjacent lake water bodies. Nevertheless the same general increases from west to east are discernible. Phillips and McCulloch (1972) in reviewing rainfall data, suggest that rainfalls over lake and over land were closely equivalent in the Lake Erie basin.

Estimates of evaporation suggests that water loss from the Lake Erie surface approximately equals the precipitation input to the same surface. Lake Erie has the highest evaporation loss of any lake in the Great Lakes system and evaporation is least in the spring and greatest in the fall and early winter.

Windrose data for January shows that winds from the west and southwesterly quadrants were dominant with related mean wind speeds of 21–26 km/h. During the spring (April) winds are much more variable in direction, and only at the east end of the lake is there a clear dominance of winds from a southwesterly direction. During this period mean speeds for dominant direction vary between 18 and 24 km/h. In July the dominant winds remain from the south and southwest, but there is perhaps a slight indication of stronger southerly component. Mean wind speeds vary between 14 and 22 km/h with higher values generally at the east end of the lake. In October southerly winds are dominant with mean wind speeds between 18 and 24 km/h; again highest speeds tend to occur towards the east end of the lake.

Lake Erie has a reputation for stormy and unpredictable weather conditions and of the 10 most severe winds all but two events have been from a generally southwest direction. D. W. Phillips (unpublished data) details the extreme wind speeds as recorded in the Erie basin, since 1918, and it is of interest to note that 70% of these events have occurred during the last 20 yr.

160 kph(100 mph)	Ship	July 4,	1969	SW
146 kph(91 mph)	Buffalo	Jan.	1950	SW
118 kph(74 mph)	Ship	Jan.	1963	NNW
117 kph(73 mph)	Ship	Nov. 14,	1972	NE
115 kph(72 mph)	Ship	Nov.	1957	WSW
115 kph(72 mph)	Long Point	Dec.	1921	SW
106 kph(66 mph)	Long Point	Jan.	1918	SW
102 kph(64 mph)	Clear Creek	Jan. 21,	1969	SW
98 kph(61 mph)	Port Colborne	Nov. 3,	1966	SW
85 kph(53 mph)	Simcoe	Feb. 16,	1967	WSW

Wind conditions over Lake Erie, where dominant direction or maximum speed tend to coincide with the main axis of the lake, produce additional complications in the form of wind wave action and seiches on the lake surface. $H_{\frac{1}{3}}$ wave heights (average of highest 33⅓% waves) in excess of 5 m are not uncommonly generated within the Eastern basin of the lake, and maximum wave heights may possibly exceed 7 m under severe conditions. Seiches of more than 2.5 m have also been reported in Lake Erie, and storm flooding and inundation pose very real threats to the lowlying shorelands, particularly at the extremities of the lake and in the areas of Long Point, Point Pelee, and the western islands.

Long-term hydrographs and precipitation records suggest that there has been an irregular but noticeable increase in precipitation in the Erie basin since about 1910. The increase has been interrupted by a number of periods of low precipitation, notably at about 1930 and again in the early 1960s. Although it is not possible to project this trend into the future, it is important to realize that such an increase is related to natural variations in climatic conditions and the influence of industrial and urban development, and agricultural practice. Population trends predict a significant increase by the year 2000 and it is reasonable to expect the atmospheric particulate load (and heat transfer) will increase somewhat proportionately; on this basis there may be some reason to expect precipitation to continue to show some slight but irregular increase. Figure 6 illustrates water level records for the Lake Erie basin since about 1860. These levels correlate, generally, with precipitation and it may be that increased particulate concentrations in the atmosphere will, through rainfall, modify long-term water levels. Such changes, however, would be very small (and in any case will be limited by outlet controls throughout the Great Lakes System).

Present approximations of flushing times in Lake Erie are: Total lake 2.5 yr, Western basin 0.2 yr, Central basin 1.5 yr, and Eastern basin 0.8 yr.

Settlement and Land Use Within the Basin

Settlement

The first known European to see Lake Erie was Louis Jolliet, who traversed the north shore in 1669; and in 1678 Father Louis Hennepin recorded the first sighting of the Niagara Falls. The earliest settlements in the area were fortified encampments, such as Pontchartrain du Detroit (Detroit) which was established by the French in 1701, to secure the fur trade. (Preparation of this section has drawn heavily upon the Encyclopedia Britannia 1969, and the Encyclopedia Canadiana 1965, as sources of reference.)

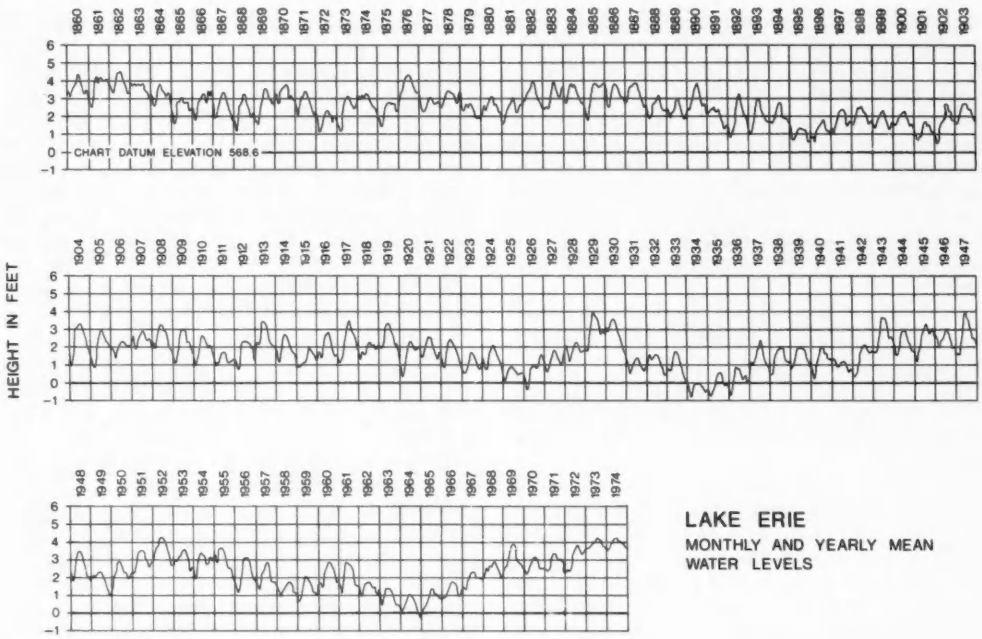

FIG. 6. Lake Erie monthly and yearly mean water levels. (After Canadian Hydrographic Service, Dep. of the Environment, Ottawa.)

During the early 1700s a few settlers moved into the region but no community development was established. Territorial possession was established by the British after the French and Indian war of 1760, and in 1763 France ceded all claims east of Mississippi to the British.

Following the close of the American Revolution the British gave up their claims to territories west of the Alleghenies, but held Detroit and continued to encourage Indian intimidation of the westward migration of settlers. After the Battle of Fallen Timbers, the Jay Treaty was signed, and the British withdrew from Fort Detroit in 1796.

Similarly, it was not until 1796 that Connecticut gave up its claim to the Western Reserve Lands, which then permitted the settlement of what is now known as the State of Ohio.

The stabilization of the political and territorial considerations at the end of the 18th century resulted in an almost simultaneous initiation of European settlement on both the north and south shores of the Lake Erie basin. It was not long, however, before the strength of resources which lay close to the southern part of the basin, and the migration of peoples through the American States to the western part of the continent, resulted in a significantly faster rate of development in that part of the basin.

In the United States, Erie, Pa., was first settled in 1753, although the town site was not laid out until 1795; Cleveland was founded in 1796, Buffalo in 1803, Toledo in 1807, and Akron in 1825.

In Canada, Sandwich (Windsor) was founded in 1796, Chatham in 1800, Talbot (St. Thomas) in 1810, London 1826, Brantford 1827, Kitchener–Galt–Preston between 1807 and 1820, and Woodstock in 1834.

It is immediately apparent that all Canadian settlements, except Windsor, are well inland on the north side of the lake while the American settlements, with the exception of Akron, are located mostly along the shoreline. This immediately provided the southern settlements with both overland and water-borne communications and transport, and a rapid development of canal systems additionally linked the Erie ports to the growing hinterland and economic markets of the central and New England states.

The Erie Canal was begun in 1817 and by 1825 it connected Buffalo with Albany and the Hudson River system; the Cleveland–Akron Canal was opened in 1827, and the Akron–Pittsburgh Canal was opened in 1840; the Toledo Canals opened to Indiana in 1843 and Cincinnati by 1845.

On the Canadian shore the first "Welland Canal" was opened between Port Robinson on Lake Erie and Port Dalhousie on Lake Ontario in 1829.

Land Use

Manufacturing industries began to gain prominence in the Erie basin in the mid-1820s, and by the early 1830s a number of American ports were also becoming important centers for trans-shipment. The coming of the railways 10–20 yr after the completion of the canals enabled a further expansion of population and trade within the basin.

A study of the development of the Detroit–Windsor region (although on the fringe of the basin) provides a good comparative example of urban and industrial developments which have taken place either side of the basin.

In 1810 the population of Detroit was about 1650 and the settlement of Sandwich (Windsor) numbered a few hundred persons. However, with the incorporation of the railways in Detroit in the 1830s and with completion of the line to Chicago in 1856, Detroit became a major shipment point for western grain and agricultural produce. It also developed the manufacturing base to serve the rapidly expanding hinterland; the first steamboat on the Great Lakes was launched in Detroit in 1818 and by the 1880s the city was the center of Great Lakes shipping traffic. By the turn of the century a broad base of manufacturing had been established including iron, steel, foundry, rail, car, boot and shoe, stove, wheel, axle, chemical, and pharmaceutical industries. In 1896 the first horseless carriage had been built by King, and by the early 1900s Durant, Chrysler, Ford, Olds, Leland, and the Dodge brothers had established the basis of today's automobile industry. Of additional industrial importance are the massive salt deposits which underlie the Detroit–Windsor area. Detroit's population grew steadily: 21,000 in 1850, 46,000 in 1860, 290,000 in 1900, 990,000 in 1920, and 1,670,000 in 1960.

The settlement of Windsor (Sandwich) grew more slowly, though it too became the focus of a number of railroad systems as a result of its advantage as an international crossing-point. Early development expanded in response to the agricultural activities of southwestern Ontario, and the Hiram Walker Distillery, established in 1858, is a good example of such an agricultural based industry. At the close of the 19th century, changing tariff agreements gave Canada preferential access to British markets and Windsor became an extremely attractive location for United States branch plant operations. The Ford plant (1913) at Windsor is typical of an early industrial development related to the influx of U.S. manufacturing. Windsor's population has grown rapidly but represents only about 10% of Detroit's at present; 21,000 in 1908, 105,000 in 1920, and 193,000 in 1966.

To the northeast of Windsor, early discoveries of oil at Petrolia led to the development of the first refineries in 1899 and the subsequent petrochemical complex of the region (Sarnia) which is now maintained by pipelines connected to western Canadian supplies.

The rapid growth and heavy and complex industrial development of many of the southern settlements in the Erie basin, the slower and less broadly based (and more agriculturally dependent) growth of the northern settlements (and western islands) typifies much of the European development of the basin.

Total populations in the Erie basin have risen from about 3.1 million people in 1900 to 11.6 million in 1970. The Canadian population remains about 10% of the total.

The development of urban and industrial land use activities in the Erie basin did not begin to take effect until about the mid-1820s and land clearance for agricultural practice was minimal. Fishing in Lake Erie was good (Christie 1974 and Leach and Nepszy 1976) and a number of small communities on both the north and south shores were quickly established. However, the building of dams for milling, the construction of canals, and the clearance of land had an early impact on stream species and earlier changes in lake fish populations were likely much influenced by these effects.

Early industry was based upon local resources as, for example, the growth of an ironworks at Erie (Pa.) which used local bog-ores from the bay swamps. Soon after woodland clearance the Canadian settlements on the north shore became widely dependent upon the import of Pennsylvanian coal, and this further encouraged the development of shipment points (such as Toledo and Erie) on the south shore.

These trends reinforced industrial activities in many south shore settlements and today they typically include iron and steel, paper, woodworking, plastic, electronics, chemical, milling, food processing, equipment, and machine industries. North of the lake the range of industrial activities developed more slowly, and early emphasis was strongly influenced by local agricultural produce and the need for agricultural equipment. Today, diversified chemical, engineering, assembly, and processing activities are widespread but because of their smaller size and more recent

development they do not appear to be so demanding upon the environment, as those of the south. Another aspect which should be recognized, is that because most of these northern centers are at some distance from the lake their immediate impact in terms of input loading is less effective and more diffuse than those situated directly on the southern shoreline.

At present most of the land in northern parts of the basin is in intensive agricultural use and, similarly, south of the lake, outside the areas of urbanization. Corn is a major grain crop, tobacco a major cash crop, and market vegetables include potatoes, lettuce, onions, tomatoes, and other varieties; fruits include apples, pears, plums, peaches, and "berries," and there are also extensive vineyards. As a result of good growing conditions and a strong demand, both cropping and husbandry have become specialized and the use of soil fertilizers and chemical controls are extensive. Agricultural contributions, as a source of input loadings, are therefore extremely important in the Lake Erie basin (Monke and Beasley 1975).

In a recent study of the Sandusky river basin (Baker and Kramer 1973), diffuse agricultural sources were seen to account for 74% of the total phosphorus contribution and, although there is a wide variation in the relative contribution of diffuse sources in the various river basins (3% in the Cuyahoga, 47% in the Maumee), it is now believed that early estimates of phosphorus loadings to Lake Erie from this source may have been too low. The character of land use, and the distance of specific input sources from the shoreline, are, therefore, of particular significance when considering the impact of human activities in the Erie basin.

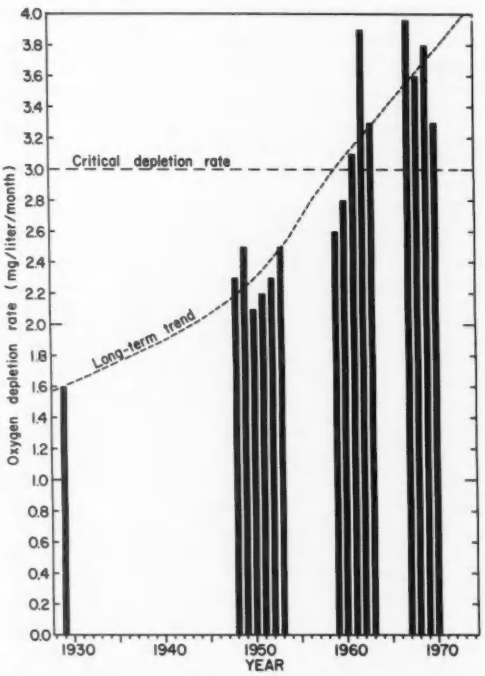

FIG. 7. Mean depletion rates for dissolved oxygen during summer in the bottom-water of central Lake Erie. Note that the critical rate of 3.0 mg/liter per month, reached in 1961, produces zero oxygen levels before the end of summer stratification. (After Dobson and Gilbertson 1971.)

Changes in Lake Erie with Time

WATER QUALITY

In Lake Erie, water quality concerns are primarily related to oxygen availability and the general excess of nutrients; water quality conditions vary markedly between the Western, Central, and Eastern basins and in all areas local effects are important. Additionally, the presence of heavy metals such as mercury, in the bottom sediments, has given rise to concerns related to such biodegradation processes (methylation) as may result in the incorporation of these contaminants in the biota. The problem of persistent organics (chlorinated hydrocarbons) and their impact on the biota as a result of bioaccumulation is also important, both to lake species and surrounding wildlife and birds.

In putting together an overview of the changes and evolution of water quality in Lake Erie, during the past 150 yr or so, it is necessary to include the documented analytical data (which mostly dates back only 60 or 70 yr), to consider the integrated stratigraphic evidence (deduced from selected sediment cores), and to develop projected loadings based upon trends in land use activities and population increase.

As most of the water in Lake Erie is derived from the upper lakes it is important to recognize this as an additional source of change as well.

Although forest clearance and improved drainage has resulted in increased sediment input loadings from some tributary sources, the severity of natural shoreline erosion and subaqueous turbulence (particularly in west and central Lake Erie) remains dominant, and it is unlikely that

there is clear evidence of a related increase in the overall concentration of suspended sediments in the receiving lake waters (see also Beeton 1969).

In Lake Erie, average concentrations of total dissolved solids have increased markedly, since records began, rising from about 140 ppm in the 1900s, to 150 ppm in the mid-1930s and to about 200 ppm by the early 1970s. The concentration of chloride ions shows a similar rate of increase, from less than 10 ppm in the early 1900s to about 25 ppm by 1970. These increases reflect the significance of increased loadings in the watershed. The fact that magnesium ion concentrations have risen little, however, implies that contributions derived from the erosion of source rocks (e.g. glacial materials and dolomitic rocks) have changed relatively little.

The historic evidence of oxygen depletion in Lake Erie is shown in Fig. 7; in this figure the development of anoxic conditions in the Central basin during the 1960s may be considered to mark the onset of severe eutrophic conditions (Dobson and Gilbertson 1971). In this figure there is also a suggestion that the rate of dissolved oxygen depletion may have increased in the 1940s (see also Carr et al. 1965). Beeton (1969), in summarizing the recent environmental changes in Lake Erie, has clearly demonstrated the differences in sediment oxygen demand in the different basins of the lake; the sediment demand rates (for 5 min) were 400 to 600 mg/g of sediment for the Western basin, 200–600 mg/g for the west part of the Central basin, 25–400 mg/g for the eastern part of the central basin, and 50–400 mg/g for the Eastern basin. The severity of the problem may also be seen in the shortening period before onset of anaerobic conditions, during summer calms.

Although evidence of reliable lake-wide dissolved oxygen data is not available before the late 1920s, it is reasonable to suggest that the Western and Central basins of Lake Erie were already highly productive (in terms of primary biomass) even before European settlement and that oxygen depletion rates in the early part of this century were similar to the early records.

External Loading

Nutrient loadings (e.g. N and P) remain a source of some considerable discussion, in regards to Lake Erie. Many early records are unreliable and the use of sediment stratigraphic data (Kemp et al. 1976) must be carefully assessed in respect to such effects as in situ degradation, bioturbation, and physical reworking. At present, best estimates appear to be based upon population growth in the basin, multiplied by appropriate coefficients.

N. M. Burns (personal communication), used published data of total nitrogen loading, of 160,969 metric tons/yr, for Lake Erie in 1966–67 (Int. Joint Comm. Rep. 1969), and related it to the basin population at this time to develop a loading coefficient of 14.4 kg/person, year of total N. Assuming a population density of 145 persons/km^2, over an area of some 77,000 km^2 these figures suggest a loading of about 2125 kg/km^2 per year; a value which falls well within the total N loading values of 1660–2810 kg/km^2 per year, developed by Vollenweider (1968), and based upon a population density of 150 persons/km^2.

However, this Lake Erie data suggests that nonagricultural inputs represent about 5.4% of the total N loading; whereas Vollenweider's data implies that nonagricultural loading would more likely represent 29–48% of the total. Although the reasons for this discrepancy are not fully understood, it is important to note that such differences exist within the total N loading.

In summary, therefore, although total N loadings appear to be within the range generally expected, nitrogen loading factors generated by this means may not properly reflect the impact of population development within the basin. Fixation of atmospheric nitrogen over the lake has not been incorporated in these estimates.

A somewhat similar approach has been used by Burns, to estimate changes in total external phosphorus loading. Three factors were considered: (i) nonmunicipal loading (agricultural and industrial), (ii) municipal (human waste), and (iii) municipal (detergent). Coefficients for the two latter factors were taken from available literature. In the development of a coefficient for factor one (i), it was also assumed that nonmunicipal phosphorus loadings varied directly in proportion to population. The coefficient was derived by the subtraction of the total municipal phosphorus load [(ii) and (iii)] from the total loading, and dividing the remainder by the population.

Municipal (detergent) phosphorus load to Lake Erie (1966–67)

Per capita use of detergent (lb/yr) (Gilbertson et al. 1972)	Population in Erie basin	Load (1966–67)
USA	$2.49 \times 9.764 \times 10^6$	$= 24.31 \times 10^6$ lb
Canada	$1.50 \times 1.397 \times 10^6$	$= 2.10 \times 10^6$ lb
	Total	$= 26.41 \times 10^6$ lb

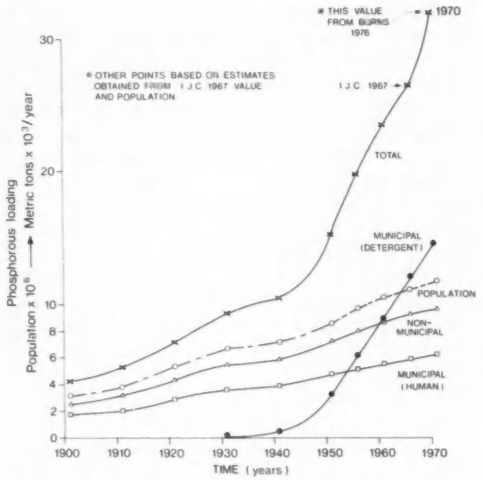

FIG. 8. Estimates of phosphorus loading to Lake Erie versus time.

Therefore, total municipal detergent load = 26.41×10^6 lb = 11.98×10^3 metric tons

Municipal (human) phosphorus load to Lake Erie (1966–67)

Human waste coefficient by approximation of Vollenweider (1968) = 540 g/person/yr

Total population Lake Erie basin (1966–67) = 11.161×10^6

Therefore, total municipal human phosphorus load = $11.161 \times 0.540 \times 10^3$ = 6.03×10^3 metric tons

Total municipal phosphorus load = $(11.98 + 6.03) \times 10^3$ = 18.01×10^3 metric tons

Total phosphorus loading to Lake Erie 1966–67 (Int. Joint Comm. Rep. 1969)

Total P loading (1966–67) = 27,299 metric tons P

Therefore, nonmunicipal loading = total P loading, less total municipal loading = 9289 metric tons P, and the nonmunicipal loading coefficient = $(9289 \times 10^6\ (g)) \div$ population = 832 g/person/yr.

The results of these calculations are shown in Fig. 8 and demonstrate the magnitude of probable increase in the external loading of phosphorus to Lake Erie.

The general concentration of trace metals in Lake Erie waters does not appear to have increased markedly since records began. However, Lake Erie sediment core data show strong evidence of increased nutrient element, trace metal, and other contaminant loadings (Kemp et al. 1976; Thomas and Jaquet 1976; Williams et al. 1976). Core data show that concentrations of organic carbon and total nitrogen have increased at a slight but continuing rate from a core date of about the mid-1700s to sometime in the 1950s. Total P exhibits a similar trend up until about the early 1940s; subsequently, however, P values rise markedly. Attendant with such increases in nutrient elements are also increases in the content of such heavy metals as cadmium, copper, lead, mercury, zinc, etc. Unpublished data (R. L. Thomas personal communication) also show recent increases of such toxic materials as pesticides and herbicides. Increases have been particularly marked since the early 1940s.

Biological Response

There is good evidence to show that the benthic communities and other aquatic forms have responded significantly to the effects of cultural impact in the Lake Erie basin, and, in fact, such changes may have begun to take place even before evidence based upon water quality parameters became available. The decline of the burrowing mayfly (*Hexagena*) from the west basin since 1930 (Carr and Hiltunen 1965), and the loss of this species as a forage item for the fish dependent upon it is a prime example of such change. Further, the spread of dense oligochaete populations (Brown 1953) in the west basin showed the impact of major point source inputs such as the Maumee River (between 1930 and 1951).

The spread of pollution tolerant forms of chironomids, etc., has become most evident in the west basin of Lake Erie, but some oligotrophic forms (such as *Mysis relicta* and *Pontoporeia affinis*) persist in the eastern part of the central basin and east basin.

The trophic status of the Great Lakes has been discussed by Vollenweider et al. (1974), and the primary productivity has been related to phosphorous loading. In their discussion the authors have classified the Western basin of Lake Erie as highly eutrophic and the Central basin as secondary eutrophic, based upon rates of primary production. This may be supported by phytoplankton biomass data, to some extent, since highest average biomass concentrations were observed in the west basin (5.3 g/m^3), as compared to Central (3.2 g/m^3), and Eastern (2.4 g/m^3) basins (M. Munawar personal communication). Details of phytoplankton biomass and species composition are further described by Munawar and Munawar (1976).

It should be noted that, in Vollenweider et al. (1974), observations of an increasing gradient of phytoflagellates from west to east also supported the concept of a decreasing trophic gradient in the same direction.

Several changes in species composition have been reported during the last 30 yr from Lake Erie, and in Munawar and Munawar (1976) several species indicative of trophic status have been reported from each of the three basins. Similar observations on zooplankton abundance and species composition, by Beeton (1969), have also suggested marked changes in the west and central basins.

Generally speaking blooms of blue-green algae began to appear in Lake Erie by the early 1960s, and phytoplankton abundance increased about threefold between 1919 and 1963.

Changes in the fish stock populations of Lake Erie (Regier and Hartman 1973; and Leach and Nepszy 1976), reflect not only the impact of degraded water quality conditions in the lake, but also changes in the flow regime of tributary streams and access to them, and the catastrophic effects of overfishing and the loss of key year-classes (Christie 1974). The impact of the sea lamprey in Lake Erie, however, has never been considered as a major contributor to the changes which have taken place. Lake Erie remains highly productive in terms of total fish catch but, during the period between 1920 and 1960, the prime stocks of whitefish, sauger, cisco, blue-pike, and lake trout were virtually eliminated. At present the fishery is dependent upon yellow perch, rainbow smelt, freshwater drum, white bass, carp, catfish, and walleye.

At the present time published works describing the effects of change upon the wetland (and marsh) ecologies of Lake Erie are negligible, and, as with the fish, it is difficult to assess the separate effects that wetland destruction, increased biocide concentrations, and increased predation may have had upon aquatic bird communities. In addition, it should be pointed out that two major flyways cross the lake and that populations of both migrant and nonmigrant species have shown declines since the turn of the century.

Summary and Conclusion

Lake Erie, southernmost in the Great Lakes system, is formed of three distinct, major, basins. The Western and Central basins are broad and shallow and only the Eastern basin may be considered to hold deep water. The general shape of the lake is strongly controlled by the regional geology, and recent glacial and postglacial events have additionally influenced the physiography of the whole basin.

The Lake Erie watershed is extremely narrow along much of the southern shore, but extends as a wide and generally flat land area to the west. Along much of the north shore morainic materials and postglacial lake deposits have produced a complex land form.

Agricultural development within the watershed is intensive, and a great variety of fruits, grain, and cash crops are grown; the use of soil fertilizers and pesticides and herbicides is heavy.

Human settlement appears to be concentrated close to the lake along much of the U.S. shoreline, whilst Canadian population centres have developed well inland from the lake. Most of the heavy industrial development has occurred along the south and west shorelines of the lake. The total population of the region is about 11 million, with about 10% in Canada.

There is evidence of a slight climatic change within the basin since the early 1900s, amounting to an increase in the annual mean temperature by about 1 C; records also suggest a slight but general increase in precipitation. Lake levels have remained generally high, although there have been notable periods of low water, associated with irregular periods of reduced inflow.

Historic evidence of cultural development within the watershed suggests that impact upon water quality was essentially localized up until the early 1900s. Sediment cores suggest that the effects of intensive urban and industrial development became widespread sometime between 1930 and 1940. These data agree well with water sample records which show that dissolved oxygen levels began to decrease even before the 1920s, and that since about 1940 the rate of degradation has rapidly accelerated.

The increase in nutrient loading to the lake from all sources has increased consistently, certainly since about 1900. Much of the increase is related to expansion of populations within the basin; however, the input of detergent phosphate (which became available sometime after 1930) has been a major cause in increasing the rate of cultural eutrophication. It is estimated that almost half of the total phosphorus input to Lake Erie was derived from this source by 1970.

Lake biota has responded to changing water quality, with pollution tolerant species now characterizing much of the Western and Central basins. Only the east basin remains essentially oligotrophic. Although water quality changes have also affected the Lake Erie communities, overfishing has had at least an equivalent impact. Lake Erie remains highly productive of fish, but

species composition has changed drastically over the past 50 yr.

Because of the complex limnology of the lake it is not yet known how Lake Erie will respond to present and future efforts to control and reduce nutrient input (as defined under the Canada–U.S. Water Quality Agreement of 1972).

Acknowledgments

I am most indebted to the many colleagues who provided me with access to their data, to references, and for their guidance and criticism in preparing this contribution. I would like, especially, to acknowledge the assistance given by N. M. Burns, A. L. W. Kemp, R. L. Thomas, and by M. Munawar. I am also indebted to A. Fraser for his assistance in the preparation of some of the data, to D. W. Phillips and J. A. W. McCulloch for unpublished climate data, and to Mr H. Dobson for the use of certain text figures.

The Director, CCIW is acknowledged for permission to publish this material.

BAKER, D. B., AND J. W. KRAMER. 1973. Phosphorus sources and transport in an agricultural river basin of Lake Erie. Proc. Conf. Great Lakes Res. 16: 858–871.

BEETON, A. M. 1961. Environmental changes in Lake Erie. Trans. Am. Fish. Soc. 90: 153–159.

1969. Changes in the environment and biota of the Great Lakes, p. 150–187. *In* Eutrophication: causes, consequences, correctives. Nat. Acad. Sci., Washington, D.C.

BERG, D. W., AND D. B. DUANE. 1968. Effect of particle size and distribution on stability of artificially filled beach Presqu'Isle peninsula. Proc. Conf. Great Lakes Res. 11: 161–178.

BROWN, E. H., JR. 1953. Survey of the bottom fauna at the mouths of ten Lake Erie, South Shore, rivers, p. 156–170. *In* Lake Erie pollution survey. Ohio Dep. Nat. Res. Div. Water Final Rep.

CARR, J. F., V. C. APPLEGATE, AND M. KELLER. 1965. A recent occurrence of thermal stratification and low dissolved oxygen in western Lake Erie. Ohio J. Sci. 65: 319–327.

CARR, J. F., AND J. K. HILTUNEN. 1965. Changes in the bottom fauna of western Lake Erie from 1930–1961. Limnol. Oceanogr. 10: 551–569.

CHAPMAN, L. J., AND D. F. PUTNAM. 1966. The physiography of southern Ontario, 2nd ed. Univ. of Toronto Press, Toronto, Ont.: 386 p.

CHIN, W. Q., AND V. YEVJEVICH. 1974. Almost periodic, stochastic process of long range climatic changes. Can. Inland Waters Dir. Sci. Ser. 69 p.

CHRISTIE, W. J. 1974. Changes in the fish species composition of the Great Lakes. J. Fish. Res. Board Can. 31: 827–854.

DOBSON, H. H., AND M. GILBERTSON. 1971. Oxygen depletion in the hypolimnion of the central basin of Lake Erie, 1929 to 1970. Proc. Conf. Great Lakes Res. 14: 743–748.

GILBERTSON, M., H. H. DOBSON, AND T. R. LEE. 1972. Phosphorus and hypolimnial dissolved oxygen in Lake Erie, p. 141–145. *In* Project Hypo, Can. Cent. Inland Waters Pap. 6.

GREAT LAKES LEVELS BOARD. 1968. Publication on shoreline physical characteristics (by county). Can. Dep. Pub. Works, Ottawa, Ont.

HOBSON, G. D., C. E. HERDENDORF, AND C. F. M. LEWIS. 1969. High resolution reflection seismic survey in western Lake Erie. Proc. Conf. Great Lakes Res. 13: 210–224.

HOUGH, J. L. 1958. Geology of the Great Lakes. Univ. Illinois Press, Urbana, Ill. 313 p.

KEMP, A. L. W., R. L. THOMAS, C. I. DELL, AND J.-M. JAQUET. 1976. Cultural impact on the geochemistry of sediments in Lake Erie. J. Fish. Res. Board Can. 33: 440–462.

LEACH, J. H., AND S. J. NEPZY. 1976. The fish community in Lake Erie. J. Fish. Res. Board Can. 33: 622–638.

MONKE, E. J., AND D. B. BEASLEY. 1975. Sediment yield and discharge from the Maumee River basin into Lake Erie. Dep. Agric. Eng., Purdue Univ., Lafayette, Ind. Unpublished manuscript, 30 p.

MUNAWAR, M., AND I. F. MUNAWAR. 1976. Lakewide study of phytoplankton biomass and its species composition in Lake Erie, April–December 1970. J. Fish. Res. Board Can. 33: 581–600.

PHILLIPS, D. W., AND J. A. W. MCCULLOCH. 1972. The climate of the Great Lakes Basin. Atmos. Environ. Serv., Dep. Environ., Toronto, Ont. Climatol. Stud. 20: 40 p.

REGIER, H. A., AND W. L. HARTMAN. 1973. Lake Erie's fish community: 150 years of cultural stress. Science 180: 1248–1255.

INTERNATIONAL JOINT COMMISSION. 1969. Pollution of Lake Erie, Lake Ontario, and the international section of the St. Lawrence River. Vol. 2. — Lake Erie. Int. Joint Comm. Rep. 189–205.

SLY, P. G., AND C. F. M. LEWIS. 1972. The Great Lakes of Canada — Quarternary geology and limnology. Int. Geol. Cong. Montreal, Que., Guideb. Trip A43: 92 p.

SLY, P. G., AND R. L. THOMAS. 1974. Review of geological research as it relates to an understanding of Great Lakes Limnology. J. Fish. Res. Board Can. 31: 795–825.

THOMAS, R. L., AND J.-M. JAQUET. 1976. Mercury in the surficial sediments of Lake Erie. J. Fish. Res. Board Can. 33: 404–412.

VOLLENWEIDER, R. A. 1968. Scientific fundamentals of the eutrophication of lakes and flowing waters, with particular reference to nitrogen and phosphorus as factors in eutrophication. OECD Tech. Rep. DAS/CS1/68-27: 95 p.

VOLLENWEIDER, R. A., M. MUNAWAR, AND P. STADLEMANN. 1974. A comparative review of phytoplankton and primary production in the Laurentian Great Lakes. J. Fish. Res. Board Can. 31: 739–762.

WILLIAMS, J.D. H., T. P. MURPHY, AND T. MAYER. 1976. Rates of accumulation of phosphorus forms in Lake Erie sediments. J. Fish. Res. Board Can. 33: 430–439.

Continuous Dynamical Computations of Water Transports in Lake Erie for 1970

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Simons, T. J. 1976. Continuous dynamical conputations of water transports in Lake Erie for 1970. J. Fish. Res. Board Can. 33: 371–384.

A numerical model of Lake Erie has been used to compute water transports throughout the 1970 shipping season on the basis of continuous wind observations from six shore stations. A vertically integrated model was used for quasi-homogeneous conditions, and a two-layer model was employed to simulate summer stratification. The computed water circulations were retained for a study of the horizontal distribution of various chemical parameters measured by regular ship cruises on the lake during this period. Computed surface elevations were compared with observed water level data to verify the model calculations and to adjust the windstress coefficient from month to month. The coefficient of correlation between observed and computed Buffalo-minus-Toledo water setup averages out to .95 for the complete integration period. In addition, a comparison was made of model results and available data from current meters maintained in the lake during this period. The results show the feasibility of quasi-operational applications of a model of the present type for computing the necessary water transport information for ecosystems modeling and water resources planning.

Simons, T. J. 1976. Continuous dynamical computations of water transports in Lake Erie for 1970. J. Fish. Res. Board Can. 33: 371–384.

Nous nous sommes servis d'un modèle numérique du lac Erié pour mesurer les transports d'eau durant toute la saison de navigation de 1970, en nous basant sur des observations continues du vent à six stations de terre. Dans des conditions quasi-homogènes nous avons utilisé un modèle intégré verticalement et, pour simuler la stratification estivale, un modèle à deux couches. Les données calculées pour la circulation de l'eau ont été retenues pour une étude de la distribution horizontale des divers paramètres chimiques mesurés lors de croisières régulières sur le lac pendant cette période. Les élévations de surface calculées ont été comparées avec les niveaux d'eau observés afin de vérifier les calculs du modèle et de rectifier le coefficient de pression du vent d'un mois à l'autre. Le coefficient de corrélation entre les valeurs observées et les valeurs calculées pour le système d'eau de Buffalo (moins Toledo) est de 0.95 en moyenne pour l'entière période d'intégration. De plus, nous avons comparé les résultats du modèle avec les données obtenues à l'aide de courantomètres maintenus dans le lac durant cette période. Les résultats démontrent qu'un modèle de ce type peut avoir des applications quasi-opérationnelles pour calculer les données sur le transport de l'eau nécessaires à l'élaboration de modèles d'écosystèmes et à la planification des ressources en eau.

Accepted November 17, 1975

Accepté le 17 novembre 1975

In recent years a new dimension has been added to the numerical modeling of large lakes by the increasing concern with the natural environment, and, in particular, the water quality of the Great Lakes. Investigations of lake dynamics date back for half a century or more, but in the past such studies were mainly concerned with oscillations of the free water surface, or when applicable, the behavior of the thermocline. In particular, the development of analytical and numerical methods of predicting storm surges in the North Sea contributed greatly to a theoretical understanding and description of such phenomena. This culminated in Platzman's (1963) comprehensive studies on the prediction of wind tides on Lake Erie. However, with the advent of water quality and ecosystems modeling, emphasis shifted from such lake level predictions to computations of water circulations and the associated transports of dissolved or suspended materials. In conjunction with this shift of emphasis, the time scales of interest changed from typical storm periods of a few days to the seasonal and yearly cycles which play a role in the management of water resources.

It is now clearly desirable to run an operational model for a system of lakes such as the Great Lakes basin on a day-to-day basis much like routine meteorological forecasts. Whereas the ultimate goal of such an operational procedure

Printed in Canada (J3992)
Imprimé au Canada (J3992)

would be to tie in the lake predictions with numerical weather forecasts, from a viewpoint of water resource planning it would not be necessary to carry out both computations simultaneously. As a first step, the lake models could be used to update the solutions for the water circulations at regular time intervals on the basis of the observed weather, thus establishing a climatological library which could serve as a framework for various water quality oriented studies.

The experiment described in the present paper forms part of a series of studies devoted to the development of such an operational model for the Great Lakes basin. In previous years, this project emphasized the mathematical–numerical aspects of the models and the verification of various experimental models (Simons 1973). In particular, a three-dimensional model of Lake Ontario has been used to simulate the lake behavior for selected episodes during the 1972 International Field Year on Lake Ontario (Simons 1974, 1975). These studies indicate that Platzman's (1963) conclusion that "a sound basis exists for operational prediction of wind tides by dynamical methods" also applies to the computation of lake circulations if our primary concern is with time scales of the order of a few days or more. These conclusions then seem to justify a move toward a quasi-operational long-term modeling effort such as the present study which is devoted to a computation of the water circulation in Lake Erie throughout the 1970 shipping season. During this period an extensive field program was carried out and numerous water quality parameters were sampled at monthly intervals throughout the Lake. In a sequel to the present paper the spatial distribution of these parameters will be studied in the light of the long-term water transports presented here (Lam and Simons 1976).

Principles of Computer Model

Numerical solutions of lake circulations and surface elevations can be obtained by step-wise time integration of the dynamical equations of motion and the continuity equation on an array of grid points. The equations include effects of the earth's rotation, pressure gradients associated with internal temperature distributions, bottom and internal friction, and horizontal diffusion of momentum. Input parameters are the bottom topography and shoreline configuration, inflow and outflow due to rivers, and the distribution of windstress at the water surface as a function of space and time. Output parameters are the values of water levels and currents in the grid points at any desired time. In view of the purpose of the present calculations, the actual output retained consisted of daily mean water transports in all grid points. Furthermore, hourly values of surface elevations in grid points adjacent to routine water level recorders were saved for model verification and once a day all variables were stored for possible restart of the computations at any one day.

In the subsequent application of the present results (Lam and Simons 1976), the assumption will be made that the lake is mixed in the vertical during spring and fall, whereas a two-layered structure is attained during summer stratification. Thus the hydrodynamic model should compute water transports on the basis of the same assumptions. For the mixed lake, it was decided to employ the familiar vertically integrated model for a hydrostatic, homogeneous, and incompressible body of water generally used in simulations of storm surges. The model adopted for the summer months consists of two layers of water with the depths and the temperatures for each layer prescribed as a function of horizontal coordinates on the basis of observations for this period. To facilitate the transition from homogeneous to stratified dynamics, the equations for the second model are formulated in terms of vertically integrated flow and the velocity difference across the thermocline. In addition to the computational advantages, this formulation will be seen to facilitate conceptual interpretation of the dynamic effects of stratification.

In the context of the present volume of papers, the principles on which these computations are based are more relevant than the mathematical formulations. Therefore, equations will be presented only as a framework for discussion, but for the complete derivations, the reader is referred to Simons (1973). Let us first consider the governing equations for the vertically integrated flow. If H denotes the local undisturbed depth of the lake, ζ the elevation of the free surface above equilibrium level, and U and V are the components of the vertically integrated water transport along the x-axis and y-axis of a coordinate system superposed on the lake surface, then the relevant equations are

$$\frac{\partial \zeta}{\partial t} = -\frac{\partial U}{\partial x} - \frac{\partial V}{\partial y}$$

$$\frac{\partial U}{\partial t} = gH\frac{\partial \zeta}{\partial x} + fV + \frac{\tau_{sx}}{\rho} - \frac{\tau_{bx}}{\rho} + A\nabla^2 U + Q_x$$

$$\frac{\partial V}{\partial t} = gH\frac{\partial \zeta}{\partial y} - fU + \frac{\tau_{sy}}{\rho} - \frac{\tau_{by}}{\rho} + A\nabla^2 V + Q_y$$

where t is time, g is the acceleration of the earth's gravity field, f is the Coriolis parameter, τ_s is the wind stress at the surface, τ_b is the bottom friction, ρ is the density of water, A is a coefficient of subgrid-scale diffusion of momentum, ∇^2 is the horizontal Laplace operator, and Q represents the effect of stratification. In keeping with common practice, nonlinear acceleration terms have been neglected and also the effects of barometric pressure gradients are omitted.

The first equation states that the water surface must rise where currents converge and vice versa. The second and third equations represent the current accelerations along the two basic coordinates as a result of the sum of the following forces and effects:

1) the pressure gradient experienced by water masses below a sloping surface;
2) the tendency of currents to deviate to the right under influence of the earth's rotation;
3) the force exerted by the wind at the water surface;
4) the bottom stress opposing the movement of bottom water;
5) the mixing of water masses with different momentum; and
6) the pressure gradients in the water resulting from internal temperature gradients and the associated density gradients.

Obviously, the last term is zero for the homogeneous lake and it would appear to be the only change caused in the vertical-mean flow upon transition to a two-layered structure. In effect, however, it turns out that this transition also affects the bottom stress and this is much more difficult to estimate in numerical terms.

In the homogeneous model, the bottom friction must be related to the vertically integrated water transports since there is no explicit estimate of bottom velocities available. Following Platzman (1963) the bottom stress is based on Ekman dynamics for shallow water, namely, $\underline{\tau}b = \rho b\underline{V}/H^2$, where b has been chosen equal to 100 cm^2/s. A complete formulation according to this theory augments the wind stress by about 25% and diminishes the pressure gradient term by a factor 5/6. Although this was not done explicitly in the present system of equations, this may be borne in mind when interpreting the windstress coefficient, used for these computations, in terms of its physical equivalent. In the layered model the bottom stress is related to the bottom velocity $\underline{V}b$ by the relationship $\tau b = \rho k\, Vb^2$ where $k = .0025$ and Vb is in practice the local hypolimnion current. As explained in the discussion of results, this tends to cause generally higher values of water transports in the two-layer model.

Next, let us consider the governing equations for the velocity difference across the thermocline. Let D_1 and D_2 be the depths of the upper and lower layer, respectively, and let the components of the velocity difference across the thermocline be denoted by u^* and v^*, then the relevant equations may be written

$$\frac{\partial u^*}{\partial t} = P_x + fv^* + \frac{\tau_{sx}}{\rho D_1} - \frac{H\tau_{ix}}{\rho D_1 D_2} + \frac{\tau_{bx}}{\rho D_2} + A\nabla^2 u^*$$

$$\frac{\partial v^*}{\partial t} = P_y - fu^* + \frac{\tau_{sy}}{\rho D_1} - \frac{H\tau_{iy}}{\rho D_1 D_2} + \frac{\tau_{by}}{\rho D_2} + A\nabla^2 v^*$$

where P is the internal pressure gradient resulting from stratification, τ_i is the interface stress at the thermocline, and all other parameters are similar to those occurring in the equations for the mean transport. The interface stress may be related to the velocity difference by the relation $\tau_i = 2\,\rho\, v\, v^*/H$ where v is the vertical eddy diffusion coefficient assumed proportional to the windstress. The pressure terms P_x and P_y include

1) the effect of a sloping thermocline;
2) the horizontal density gradients in each layer; and,
3) the time rate of change of thermocline depth.

The mathematical formulations are not relevant in the context of this discussion, but are available from the author.

In general, the thermocline depth is not known and hence another equation is needed to complete the system. This equation is similar to the first equation for the vertically integrated flow. However, unlike the free surface, the thermocline is not necessarily impermeable and thus we must allow for a vertical water transport across this imaginary surface. Clearly, any local convergence of hypolimnion currents can be balanced by either an increase in hypolimnion thickness or an upward flux of water through the thermocline. At this point, an arbitrary assumption must be made regarding the degree of permeability of the interface. In conventional two-layer models a thermocline is treated as an impermeable interface, but for long-term computations this is obviously not acceptable. As an alternative, let us consider the following procedure for the present experiment.

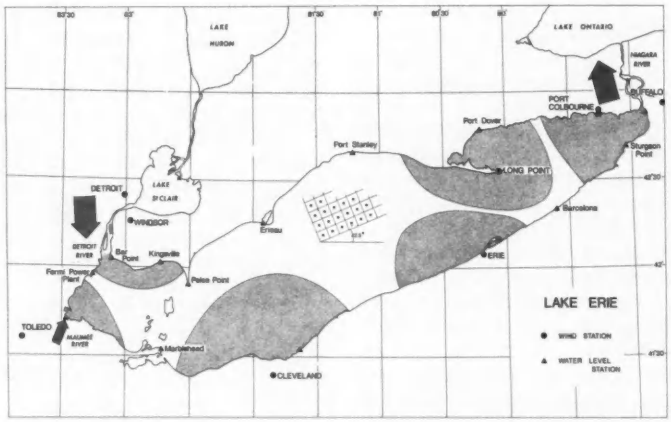


FIG. 1. Lake Erie wind and water level stations, major rivers indicated by arrows, and influence areas of wind stations indicated by shading. Numerical grid shown in center of map.

As a first approximation, the thermocline depth as well as the temperatures will be obtained by interpolation between observations which are available from ship cruises at monthly intervals. The procedure would be meaningless if we were interested in short-term phenomena, such as internal waves, but on the other hand it is known that the so-called internal radius of deformation of the thermocline is less than the grid distance used in this study and thus we cannot hope to simulate thermocline deflections. In essence, then, the assumption is made that the interpolated stratification data are adequate to evaluate the internal or "baroclinic" pressure gradients for the time scales of interest here. However, it must be anticipated that this approximation will result in overestimates of upwelling and downwelling since the model lake has no built-in resistance against vertical circulations. We will return to this when discussing the results of computations.

Computational Procedures

This section will be concerned with computational details and model input and the information presented herein is not essential for an understanding of the discussion of results. The numerical calculations were performed on two Richardson lattices conjugate in space but coincident in time (Platzman 1963). In order to reduce grid dispersion, the lattices are oriented under an angle of 45° with respect to the longitudinal axis of the lake as discussed by Simons (1973). The resulting grid appearance is indicated in the center of Fig. 1 where the corners of the squares represent velocity points and the surface elevations are defined in the centres of the squares. The transport components are computed at integer multiples of the timestep whereas the water levels are half a timestep out of phase. A grid element adjacent to the shore may be either a regular square or a half square depending on the local shape of the boundary. For the time-stepping, centered differences are used for the divergence terms and the pressure gradient, a forward scheme is applied to the bottom friction and diffusion terms, and the Euler Trapezoidal rule is used for the Coriolis terms. The complications associated with the implicit character of the latter are eliminated by solving the system of two simultaneous equations for the transport components to arrive at an explicit algorithm.

From a viewpoint of computational stability, the two most relevant conditions, with regard to the vertically integrated transport equations can be stated approximately as follows (see e.g. Platzman 1963)

$$\Delta t < \frac{2H_{min}^2}{b}; \quad \Delta t < \frac{\Delta s}{\sqrt{gH_{max}}}$$

where Δs is equal to the sides of the grid squares shown in Fig. 1 and b is the bottom stress coefficient defined above. The first condition results from the action of the bottom friction, the second is imposed by the free surface waves. In the present model $b = 100$ cm²/s and $\Delta s = 6.667$ km, the shallowest depth in any one grid point is 2 m and the maximum depth is 62 m, thus Δt must be less than 4.5 min. This means that 1 s of com-

puter time on a CDC-6600 allows for a prediction of about 2 h of real time. At commercial rates this translates to a cost of about $5.00 for 1 day of computation, which seems quite acceptable from the viewpoint of quasi-operational application of the model.

Since the equations for the velocity difference across the thermocline do not contain the free surface gradient, the computational timestep is not restricted by the same considerations. For the present calculations a timestep of 15 min was used. In view of this longer time step and because of the fact that the internal pressure gradients are obtained from observations, the shear-equations add only a fraction of the computer time required by the vertically integrated equations.

Let us next consider some aspects of the input data for the model. The windstress field over the lake is obtained by interpolation from surface wind observations at meteorological stations on the periphery of the lake. Stations which can be considered for this purpose are Windsor, Ont.; Detroit, Mich.; Toledo and Cleveland, Ohio; Erie, Pa.; and Buffalo, N.Y.; all of which are airport stations. In addition, hourly wind reports are available during the shipping season for Long Point and Port Colborne Light in Ontario. The geographical distribution of the stations shown in Fig. 1 is such that two of the stations, Buffalo and Detroit, can be discarded, thus leaving six stations to derive the wind field over the lake. The interpolation formula used for this purpose is of the form

$$\tau(x,y,t) = \sum_n \omega_n(x, y)\tau_n(t)$$

where ω_n is a dimensionless weighting function and τ_n is an effective surface stress over water as derived from the wind observed at station n.

The stress is assumed to be related to the wind W_n by the usual quadratic stress law

$$\tau_n = k_n\rho_a|W_n|\underline{W}_n$$

where ρ_a is the density of air and k_n is a skin friction coefficient which in principle varies from one station to the next to account for obstacle effects, anemometer elevation, and differences in roughness between land and water. As suggested by Platzman (1963) it would be useful to write the latter as $k_n = \gamma_n k_o$ where k_o is a representative coefficient for over-lake conditions and the factor γ_n reduces all stations to a common exposure. The basic coefficient k_o itself must be expected to vary in time as a function of stability, i.e. the difference between air and water temperatures, and possibly as a function of windspeed (indicating a deviation from the quadratic stress law). Estimates of the stress coefficient vary over a wide range, and apparently the values depend strongly on the manner in which they are determined (see e.g. Bruce et al. 1968). Following Platzman (1963) the procedure adopted for this experiment was to ignore variations of the exposure coefficient γ_n and to adjust the stress coefficient k_o a posteriori by comparison of computed and observed water setup, and thus all computations were made for a uniform stress coefficient $k_o\rho_a/\rho_o = 3 \times 10^{-6}$ where ρ_o is the density of water. Recalling the earlier statement that a complete formulation of the shallow water Ekman friction augments the windstress by 25% and noting that the ratio ρ_a/ρ_o is about 1.25×10^{-3}, the corresponding windstress coefficient would be of the order of 2×10^{-3} for the homogeneous model.

The weighting function $\omega_n(x,y)$ is based on the simple assumption that the influence of each station in a given gridpoint is inversely proportional to some power of the distance. The normalized influence function is then

$$\omega_n(x,y) = r_n^{-\beta} \Big/ \sum_m r_m^{-\beta}$$

where r_n is the distance between point (x,y) and station n. The exponent β was chosen by visual inspection and comparison of interpolated maps for a range of exponents, which led to a choice of $\beta = 2.0$, the same value as the one adopted by Platzman (1963). Following Platzman, the shaded areas in Fig. 1 indicate the domains of predominant influence, as measured by the contours $\omega_n = 0.5$, for all stations.

In addition to the windstress, the model requires input data on the water supply to Lake Erie. The main tributary inputs to Lake Erie are the discharges from the Detroit River and the Maumee River at the western end of the lake. Together these river inputs amount to about 200,000 cfs, out of a total water supply of about 215,000 cfs from all rivers and distributed runoff. Estimates of precipitation and evaporation indicate a net evaporation of the order of 5,000 cfs, thus leaving a river outflow of about 210,000 cfs. This outflow takes place through the Niagara River and the Welland Canal at the eastern end of the lake. Since the effects of the distributed sources on the lake circulation can be safely neglected, the only river inputs included in the present experiment are those from the Detroit River and the Maumee River. The outflow through the Niagara River is assumed at all times to balance these water inputs, and the outflow through the

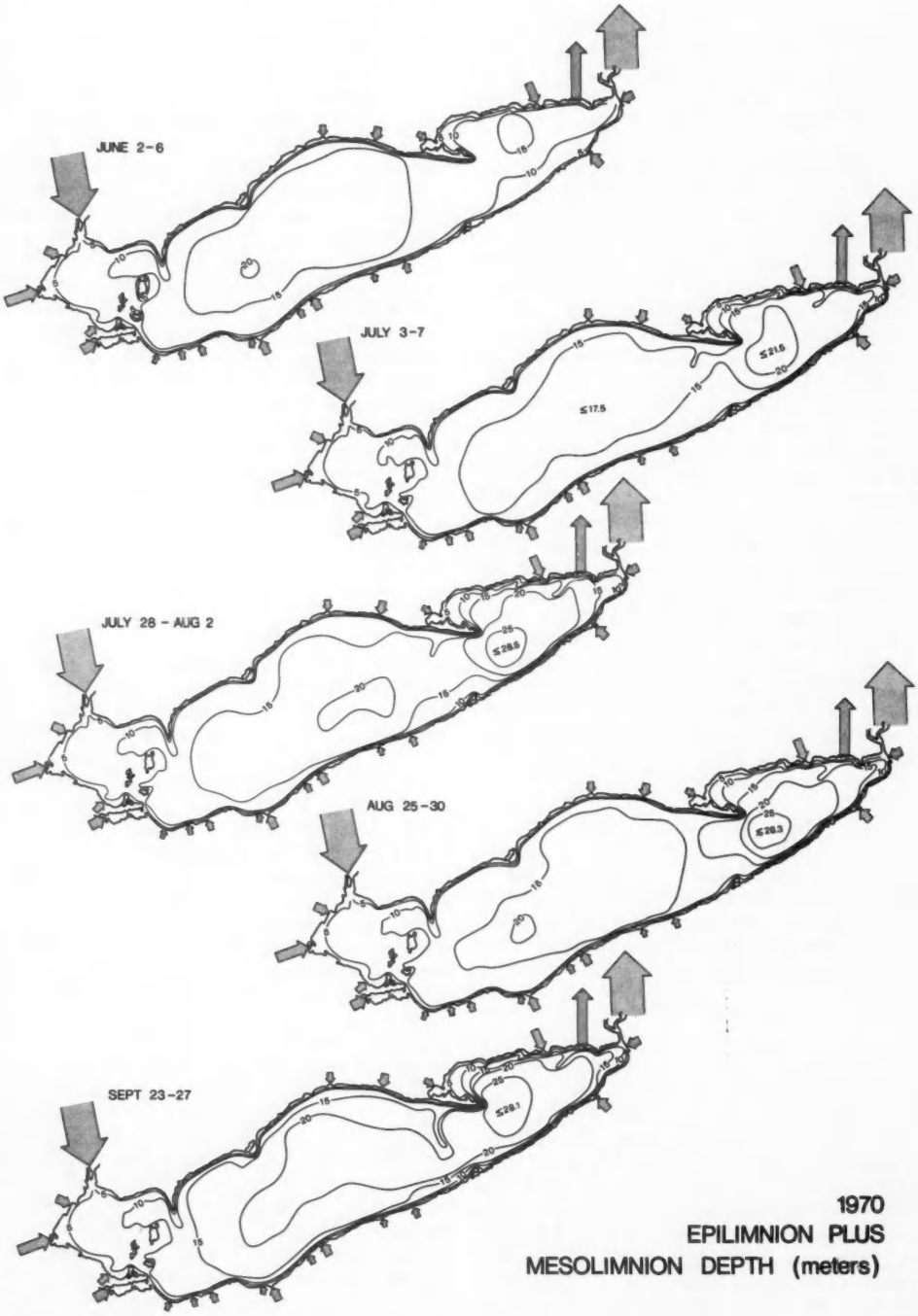


FIG. 2. Observed thermocline depths at approximately monthly intervals during the summer of 1970.

Welland Canal is neglected together with the exclusion of the distributed sources. The river discharges are specified in the boundary velocity points adjacent to the junctions of the rivers and the lake indicated in Fig. 1. Note that from a numerical viewpoint it is essential that each river discharge be distributed equally over both lattices of the grid. This is accomplished by simply prescribing that the directions of the river flows must coincide with either of the two coordinates of the grid indicated in Fig. 1.

Results of Computations

The vertically integrated model was run continuously for the 1970 shipping season, i.e. April through December 1970. From June through September, the two-layer model was run in addition to the vertically integrated model. In terms of the earlier discussion of model principles, this meant that,

1) the additional internal pressure gradients were included in the equations for the vertical mean flow;
2) the bottom stress was evaluated from hypolimnion velocities; and
3) the equations for the velocity difference across the thermocline were added.

The observed thermocline depths used to evaluate the internal pressure gradients are shown in Fig. 2, and the temperature gradients were computed from observations analyzed by N. M. Burns (personal communication).

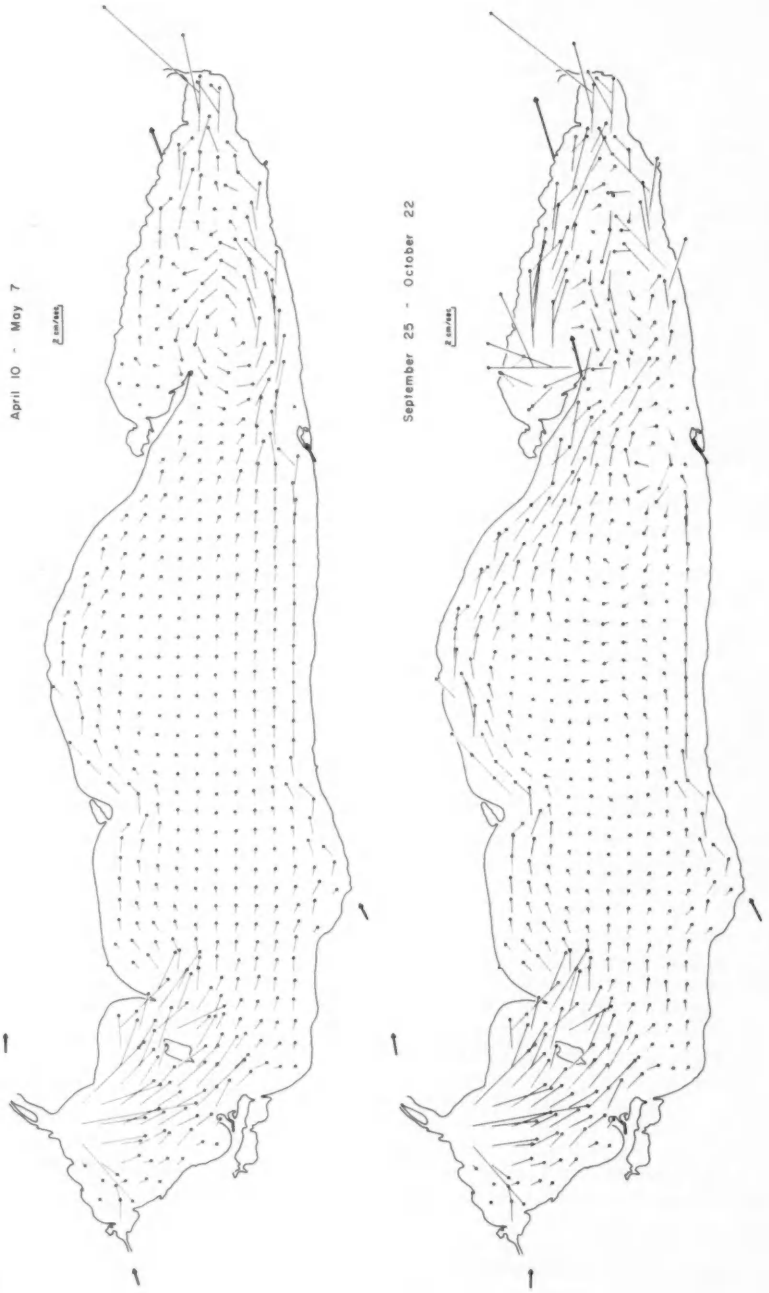
Hourly values of surface elevations in gridpoints corresponding to routine water level recorders were saved on magnetic tape together with daily-mean transport values in all gridpoints. The former were used for model verification whereas the latter will be used to compute transports of chemical parameters in a continuation of this study (Lam and Simons 1976). Since the ship cruises which measured these parameters were scheduled at intervals of four weeks, the total integration period was divided into intervals corresponding to the cruise spacings. The dates of the cruise intervals are shown in the second column of Table 1. The numerical computations started on April 1, whereas the first cruise interval starts on April 10. It is, therefore, reasonable to expect that the artificial initial conditions of no motion imposed on the model, will not exert much influence on the computations.

A comparison of observed and computed water levels can be used as a basis for verification of the model computations, and, in particular, can serve to adjust the windstress coefficient so as to effect a better agreement between observations and model results. Since the one-layer model is linear, the solutions for the water levels and the transports will be directly proportional to the stress coefficient. Fifteen water level recorders were operated during 1970 by the U.S. Lake Survey and the Canadian Hydrographic Service (see Fig. 1). The differences in water levels measured at opposing ends of the lake are related to the corresponding wind component and hence are ideally suited for model verification. Probably the most sensitive criterion in this regard is the wind setup between Buffalo and Toledo at opposite ends of the longitudinal axis of the lake. Comparison of day-to-day variations of the observed and computed Buffalo-minus-Toledo setup for each of the cruise intervals indicates a general agreement between observations and model results. A more quantitative estimate can be obtained by statistical procedures.

TABLE 1. Comparison of observed and computed Buffalo-minus-Toledo water setup. r = correlation coefficient, S = slope of linear regression line from observed on computed setup. Numbers in parentheses refer to two-layer model.

28-day period		Correlation	Slope
Number	Dates	r	S
1	Apr. 10–May 7	.93	.94
2	May 8–June 4	.95	.85
3	June 5–July 2	.97 (.95)	1.02 (.91)
4	July 3–July 30	.95 (.93)	.93 (.81)
5	July 31–Aug. 27	.93 (.95)	.90 (.79)
6	Aug. 28–Sept. 24	.97 (.96)	1.02 (.90)
7	Sept. 25–Oct. 22	.96	1.35
8	Oct. 23–Nov. 19	.96	1.09
9	Nov. 20–Dec. 17	.92	1.29

Table 1 represents some statistical parameters concerning observed and computed Buffalo-minus-Toledo water setup for each of the 24-day cruise intervals. The numbers in parentheses for the summer months represent results from the two-layer model. The correlation coefficients shown in the third column attest to a nearly linear relationship between observed and computed set-up. The slope coefficient of the linear regression equation for observed versus computed set-up is shown in the next column. There is a tendency for this coefficient to increase with time which must be related to decreasing stability associated with air–water temperature differences. However, in only one third of all homogeneous computations, the value deviates from unity by more than ten percent. In order to effect agreement between observed and computed values in these cases, the windstress coefficient used for the homogeneous


April 10 - May 7
2 cm/sec
September 25 - October 22
2 cm/sec

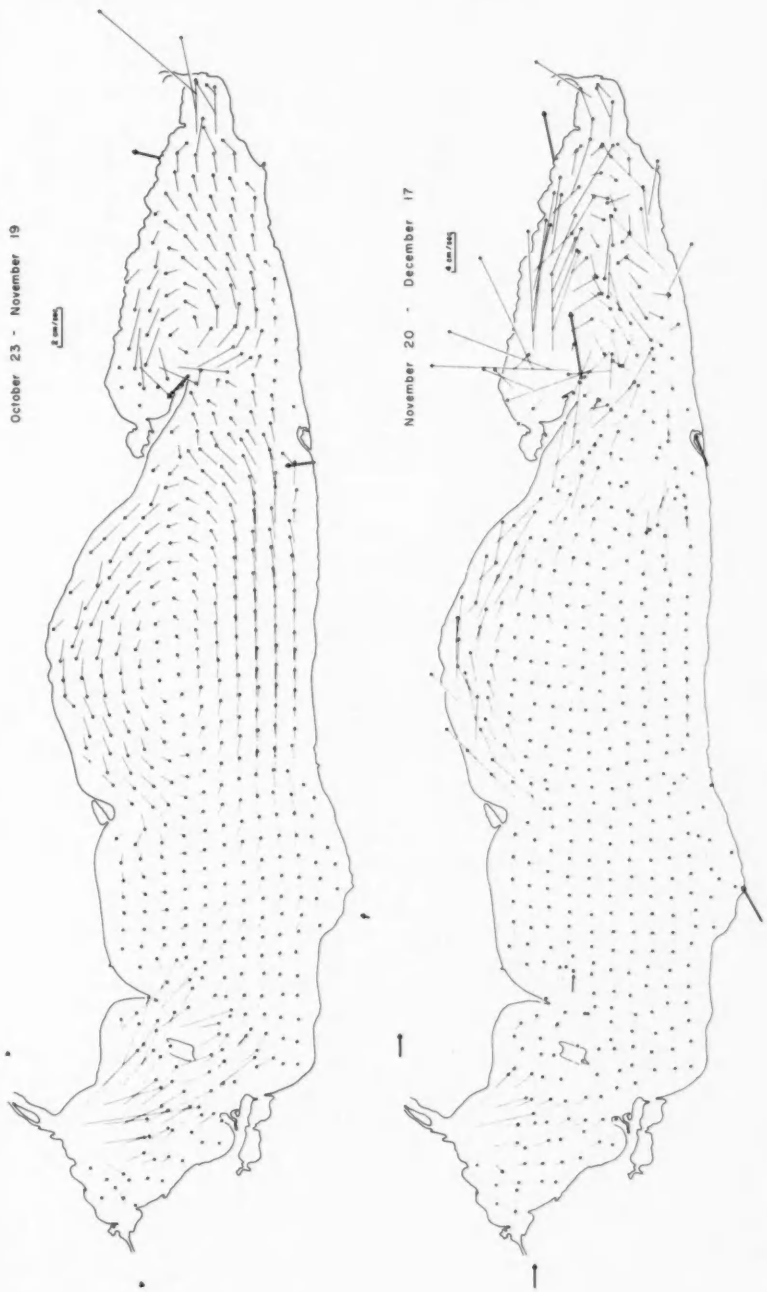
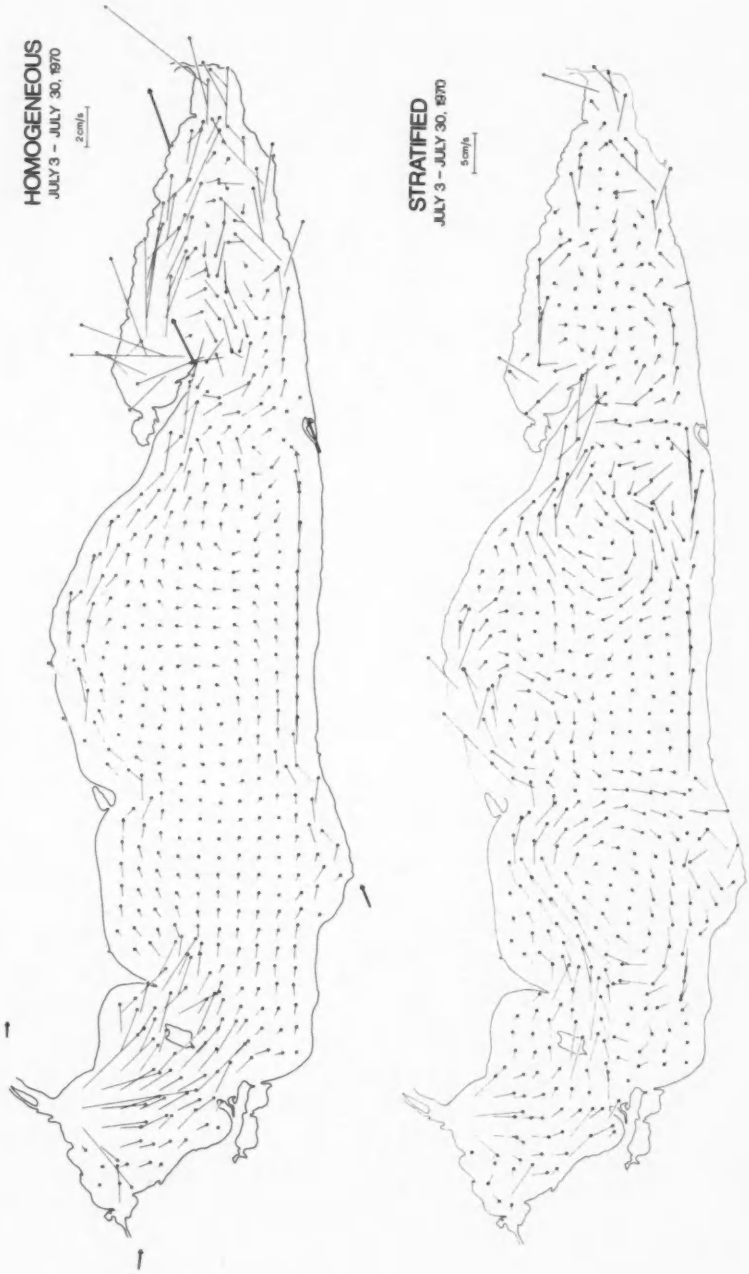


FIG 3. Depth-averaged Lake Erie water circulations computed by homogeneous model for cruise intervals indicated by the dates above each map. Heavy arrows indicate directions and relative magnitudes of windstresses.


HOMOGENEOUS
JULY 3 – JULY 30, 1970
2cm/s
STRATIFIED
JULY 3 – JULY 30, 1970
5cm/s

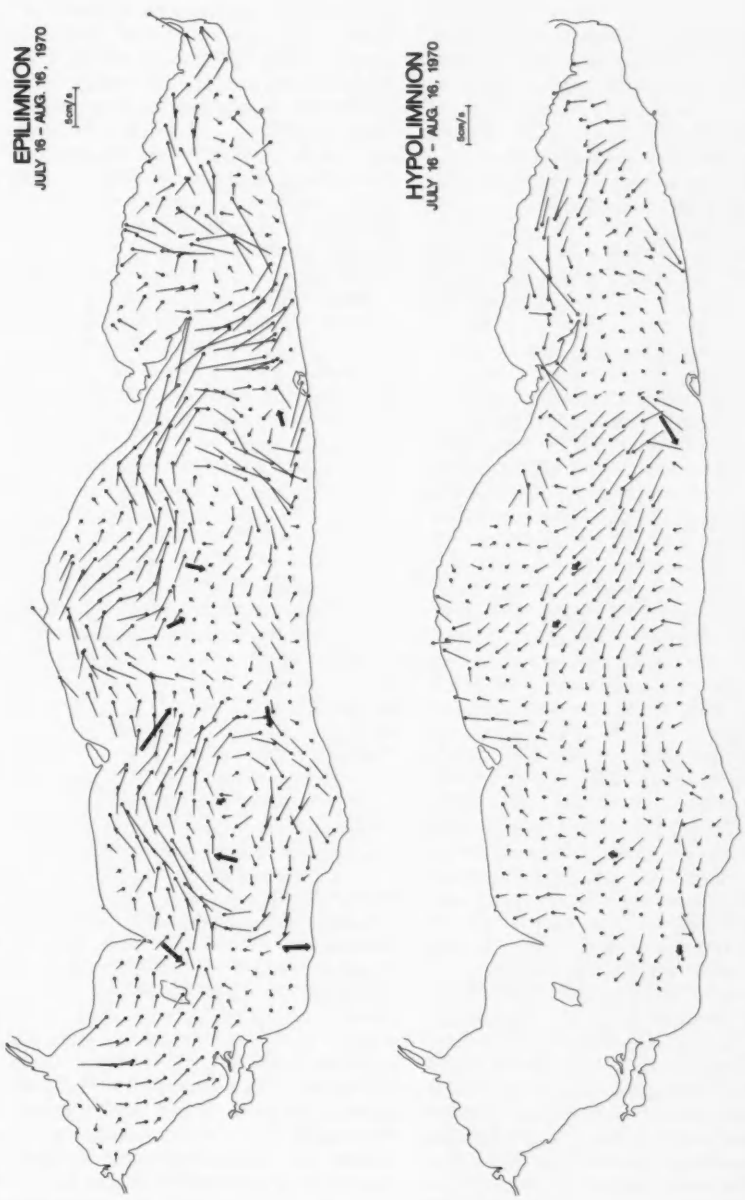


FIG. 4. Computed Lake Erie water circulations for indicated periods. Two upper maps show depth averaged currents computed by homogeneous and two-layer model, respectively. Lower maps show computed epilimnion and hypolimnion currents together with observed currents (heavy arrows).

model should be reduced for the second period and increased for period 7. In other words, the computed water circulations should be multiplied by the numbers presented in the last column of Table 1 in order to obtain the best estimate of the actual currents. It is also seen that the two-layer model requires a smaller windstress coefficient or, conversely, that the currents obtained from the summer model should be considerably reduced. This difference is related to the bottom stress mentioned under "Principles of computer model," but a detailed discussion would go beyond the context of the present paper.

Proceeding to the presentation of computed currents, we must limit ourselves to a few representative results. In view of the subsequent application of these currents (Lam and Simons 1976) it appears to be of most interest to look at current patterns averaged over cruise intervals. First, Fig. 3 shows the depth-averaged currents obtained from the homogeneous model for spring and fall. The heavy arrows outside the lake represent the corresponding mean winds. Although the period from May 8 to June 4 was also treated as a fully mixed period, the resulting map appears identical to the first one and is not displayed. Coincidentally, the chloride data for the intermediate cruise are missing and hence the subsequent computation of chloride distributions combines the two periods into a single one.

It is seen that the water circulation in the shallow Western basin is largely dominated by the discharge from the Detroit River and thus appears similar for all periods. The circulation pattern in the Central basin during spring and early fall is typical for a wind blowing parallel to the lake axis. In the shallow shore zones, the water moves in the same direction as the wind, whereas in the interior of the basin the water runs against the wind or the currents are very weak. An excellent discussion of this effect has recently been presented by Bennett (1974). The reason for this behavior of the currents is essentially that the same wind must move a larger mass of water in the deeper portions of the lake than in the shallower zones, and, consequently, the water along the shores tends to be accelerated more in the direction of the wind. The circulation in early November shows a remarkable counter-clockwise gyre throughout the Central basin, which is clearly related to the somewhat unusual wind distribution. In early December, the circulation is again back to normal but now there is a distinct return flow in the Central basin because the winds were very strong throughout this period and thus the underlying river-hydraulic flow is completely obscured. Note that the scale of the current vectors has been reduced by one-half in the last map.

The four stratified periods, June through September, show considerably less variation from one month to the next because average wind conditions varied little and the general topography of the thermocline was rather well maintained as shown in Fig. 2. It would appear most meaningful, therefore, to select the results for a particular period and discuss these in some detail. The upper maps of Fig. 4 show computed depth-averaged currents for the month of July as obtained from the homogeneous and the stratified model, respectively. Notice that the currents are plotted on different scales in the two maps. Thus in the Western and Eastern basin the results are essentially the same, whereas in the Central basin the mean currents in the two-layer model are much larger than in the homogeneous model. These larger mean transports are mainly caused by the fact that the hypolimnion water tends to run against the wind and hence the frictional effect of the bottom on the total water column tends to act in the same direction as the wind itself. On the other hand, in the one-layer model the bottom stress acts against the vertically integrated flow and thus it will always retard the water circulation, in particular, in the shore regions where the largest velocities occur. It should be noted that this effect is much more pronounced in time-averaged circulations as shown here than in day-to-day current patterns. Furthermore, it is seen that the character of the stratified solution is similar to the homogeneous solution except for the southwestern corner of the Central basin. In retrospect, it might have been better to select a smaller coefficient of bottom friction for the homogeneous model, which would have resulted in larger mean water transports.

The lower maps of Fig. 4 show model solutions for the epilimnion and hypolimnion separately. The selected period is not one of the regular cruise intervals but it has been chosen for comparison with observed lake currents. During the summer of 1970, a number of current recorders were operated in Lake Erie by U.S. and Canadian agencies. The majority of the metered current records started on July 14 and ended between August 18 and 27. Thus, the observed currents were averaged from July 16 to August 16 and are shown as heavy arrows in the maps of computed currents averaged over the same period. The epilimnion current meters were generally located about 10 m below the surface, while the hypolimnion recorders were placed at twice this depth. The thermocline depth in the Central basin is approximately 15 m (Fig. 2).

There is reasonable agreement between observed and computed circulation patterns in the interior of the basin, in spite of the fact that the computed

currents are depth-averages for each layer. As expected, this agreement deteriorates at the edges of the thermocline. For example, the two current recorders in the southwest corner of the basin were located at 10 and 13 m below the surface while the local mean thermocline depth was 11.5 m, a few meters above the bottom, thus the upper current meter is not representative of the epilimnion transports. The observed hypolimnion current in the southeast corner appears worse than it actually is, if one takes into account that the coarse grid tends to shift the coastal boundary layer away from the shore. In fact, the computed hypolimnion currents tend to be westward in all but the first row of gridpoints. An interesting aspect of these maps appears to be that observed currents seem to generally fit into the computed pattern but seldom agree with the result for the adjacent gridpoint. This makes it very difficult to compare observed time series of currents with their counterparts in any particular gridpoint. Another pronounced result is the complex flow structure computed by the model. It is obvious that an extremely large number of current meters would be required to actually document such flow patterns. In spite of the complicated nature of the computed circulations, there are some dominating features. Thus, the epilimnion displays a distinct preference for clockwise gyres. To some extent this appears also in the homogeneous maps, but the effect is much stronger in the stratified model. This result is a direct consequence of the use of observed thermocline depths in our model. It is well-known in meteorology and oceanography that winds and currents tend to increase with height if the air or water to the right is warmer than to the left of a person looking downstream. Figure 2 shows that the epilimnion is deeper in the interior than in the shore zones and thus the upper layers of the lake would tend to run around the central basin in a clockwise sense. Note, however, that our computations do not establish why the thermocline likes to adopt this particular configuration.

In concluding this presentation of results, a few remarks should be made on the hypolimnion currents. The computed northwesterly flow is supported by the present observations as well as other field data. However, as indicated at the end of the section on "Principles of computer model," our model overestimates this flow because the water is allowed to flow freely across the thermocline. In reality, the thermocline will move upward in upwelling regions and downward in downwelling zones. As a result, pressure gradients are created in the hypolimnion which tend to oppose further thermocline displacements. This inherent stability is missing here and hence the vertical circulations are exaggerated. As another extreme, we could visualize an impermeable thermocline and thus essentially no vertical circulation over time periods of the order of 1 mo. There is a simple way of dividing the total currents in two components, namely, a divergent component related to vertical motions and a nondivergent component without any upwelling or downwelling. By dropping the first component one obtains a solution for an impermeable thermocline, and any amount of vertical mixing may be included by adding part of the divergent component. This procedure is obviously completely artificial but it appears of some interest and will be explored briefly in the chloride computations (Lam and Simons 1976).

Summary and Conclusions

In this study an attempt was made to employ recently developed numerical models of water circulations for the purpose of establishing the transport framework for water quality models. Whereas the circulation models were originally developed in the context of investigations of relatively short-term physical phenomena such as storm surges, it is shown herein that it is technically and economically feasible to apply the same techniques for the longer time scales associated with ecosystems studies. Numerical computations were made of the water circulation in Lake Erie throughout the 1970 shipping season, with the use of a vertically integrated free-surface model. In addition, separate epilimnion and hypolimnion circulations were computed from June through September from a simplified two-layer model utilizing observed distributions of temperatures and thermocline depths.

Since the application of the computed currents in studies of material transports (Lam and Simons 1976) deals with time scales of days and longer, it is not necessary to employ a free surface model with the inherent short computational time step. However, an advantage of these models is that they predict water levels and it is well known that an extensive observational network exists to document water elevations on a routine basis. Consequently, this part of the results can be easily verified and it can be used to tune important adjustable parameters of the model such as the windstress coefficient. Comparisons of computed and observed water elevations in the present study gave very high correlation coefficients. This would indicate that the model is basically a sound replica of the real lake.

The proper modelling of stratified lakes is still a problem that requires a great deal of further investigation. In this study, an effort was made to

include the major effects of temperature stratification on the internal current distributions without attempting to simulate the complete dynamics of a stratified lake. In view of the availability of stratification data at monthly intervals and the fact that the general stratification pattern was very similar throughout the summer, a model was constructed to utilize this information. Although this two-layer model has a number of shortcomings, as discussed in the text, it at least incorporates some basic concepts regarding stratified lake dynamics and it is, by definition, tied to observed density distributions. Furthermore, comparison of computed currents with available data indicates a basic agreement between model and prototype.

In general, the computed water circulations are consistent with our knowledge of the currents in Lake Erie, a summary of which has been presented by Hamblin (1971). It should be realized that the majority of the available material is qualitative in nature and does not allow us to distinguish seasonal changes or effects of variable winds. With regard to the shallow Western basin, the observations clearly establish the dominant influence of the Detroit River and there is considerable evidence that the larger portion of the outflow takes place to the north of Pelee Island. At times, some recirculation has been noticed to the south of the island. Our computations show a similar pattern but the return flow (Fig. 4) does not propagate into the Western basin. Of course, the maps presented here are monthly mean patterns and do not exclude the possibility of observing different phenomena on a day-to-day basis.

Much of our knowledge of the water circulation in the Central basin is derived from a field program in 1963–64, organized by the U.S. Federal Water Pollution Control Administration. Drogues, seabed drifters, dye, and mechanical current meters were employed, but it is clear that the movement of drifting objects or data from a sparse network of recording equipment can be properly interpreted only if the circulation pattern is sufficiently simple. Instead, our computations point at a complicated system of semi-independent cells or gyres. Nevertheless, the basic conclusions are the same for the observational and the modelling study. Along the south shore, a fairly narrow eastward drift is a quite common feature of the Central basin circulation (Fig. 3). The flow along the northern shore appears to take part in circulation cells extending throughout the basin, the direction of which is controlled by wind and stratification. In summer, surface waters tend to move away from the north shore and returning hypolimnion waters create a zone of predominant upwelling along the northwestern shore of the central basin (Fig. 4). These mutually consistent circulations derived from field programs and computer models appear to constitute an adequate base for computing the movement of dissolved or suspended material in the lake.

Acknowledgments

The thermocline depths of Fig. 2 were analyzed by N. M. Burns. The observed currents in Fig. 4 were collected during the joint Canadian–U.S. Project HYPO and were processed by E. B. Bennett.

BENNETT, J. R. 1974. On the dynamics of wind-driven lake currents. J. Phys. Oceanogr. 4: 400–414.

BRUCE, J. P., R. K. LANE AND H. S. WEILER. 1968. Processes at the air–water interface. Proc. Conf. Great Lakes Res. 11: 268–284.

HAMBLIN, P. F. 1971. Circulation and water movement in Lake Erie. Dep. Energy, Mines Resour. Ottawa, Canada. Sci. Ser. No. 7: 50 p.

LAM, D. C. L., AND T. J. SIMONS. 1976. Numerical computations of advective and diffusive transports of chloride in Lake Erie, 1970. J. Fish. Res. Board Can. 33: 537–549.

PLATZMAN, G. W. 1963. The dynamical prediction of wind tides on Lake Erie. Meteorol. Monogr. 4: 44 p.

SIMONS, T. J. 1973. Development of three-dimensional numerical models of the Great Lakes. Can. Cent. Inland Waters, Sci. Ser. 12: 26 p.

1974. Verification of numerical models of Lake Ontario, Part1: Circulation in spring and early summer. J. Phys. Oceanogr. 4: 507–523.

1975. Verification of numerical models of Lake Ontario, Part 11: Stratified circulations and temperature changes. J. Phys. Oceanogr. 5: 98–110.

Surficial Sediments of Lake Erie

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On the basis of extensive echosounding and grab sampling, three major units have been recognized in Lake Erie: till and bedrock, glaciolacustrine clay, and postglacial muds. These units represent the late glacial and postglacial evolution of the basin and occur in an offshore younging sequence. The main basin of the lake is subdivided by residual glacial moraines into four depositional basins: Western, Sandusky, Central, and Eastern basins. The sediment texture has been defined by moment measures (mean, standard deviation, skewness, and kurtosis), the trends of which are related to the mixing of two primary grain-size populations in the sand- and clay-size ranges. A third grain-size mode in the silt size, composed of fine quartz with some carbonate, has been recognized. This mode has a modifying effect on the symmetry of the two primary populations and may, to some extent, be sufficiently abundant to behave as a discrete population. The trends in the textural characteristics, particularly skewness and kurtosis, have been utilized to define energy regimes at the sediment–water interface which indicates three distinct sedimentary or hydraulic regions: 1) Western basin region — Fine-grained sediment accretion in shallow water related to an imbalance in sediment budget, with high input loadings of fine-grained sediment, and deficit in coarse materials, with an excess of input over sediment export to the Central basin region. This results in net sediment accretion in shallow water with texture in disequilibrium with environmental energy, which produces mixing and suspension, followed by redeposition; 2) Central basin region — West to East coarsening of sediment in textural equilibrium with hydraulic energy, as it relates to increasing fetch under westerly and southwesterly prevailing winds; 3) Eastern basin region — Deepwater basin with sediments showing decreasing size offshore with increasing water depth. The deepwater sediment is modified by the influx of substantial quantities of the silt-size material derived from shoreline erosion in the north shore of the Central basin region.

The interrelationships of parameters indicate textural dependence on mineralogic composition, particularly important being the relationship of clay concentration to mean grain size. This has particular value in modelling the physical behavior of clay-associated geochemical elements such as phosphorus.

THOMAS, R. L., J.-M. JAQUET, A. L. W. KEMP, AND C. F. M. LEWIS. 1976. Surficial sediments of Lake Erie. J. Fish. Res. Board Can. 33: 385–403.

D'abondants sondages à écho et échantillonnages à la benne nous ont permis d'identifier trois unités majeures dans le lac Erié: argile à blocaux et socle rocheux, argile glacio-lacustre et vases postglaciaires. Ces unités reflètent l'évolution fin glaciaire et postglaciaire du bassin et se trouvent en une succession de plus en plus jeune vers le large. Le bassin principal du lac est séparé par des moraines glaciaires en quatre bassins de déposition: les bassins ouest, Sandusky, central et est. La texture du sédiment a été définie par des mesures de moments (moyenne, écart type, obliquité et aplatissement), dont les tendances sont liées au mélange de deux populations primaires de grosseur de grain dans la gamme du sable à l'argile. Nous avons identifié un troisième mode correspondant à la grosseur des particules de limon, composé de quartz fin mélangé avec un peu de carbonate. Ce mode modifie la symétrie des deux populations primaires et peut, jusqu'à un certain point, être suffisamment abondant pour se comporter comme une population distincte. Les tendances dans les caractéristiques de texture, surtout l'obliquité et l'aplatissement, ont été utilisées pour définir les régimes énergétiques à l'interface sédiment–eau qui montre trois régions sédimentaires ou hydrauliques distinctes: 1) Région du bassin ouest — Accrétion de sédiment à grain fin dans les eaux peu profondes, liée à un déséquilibre du budget sédimentaire, avec apport

Printed in Canada (J3993)
Imprimé au Canada (J3993)

abondant de sédiment à grain fin et déficit en matériaux plus gros, et surplus des apports par rapport aux exportations vers la région du bassin central. Ceci entraîne une accrétion nette de sédiment dans les eaux peu profondes, avec texture en déséquilibre avec l'énergie ambiante, ce qui produit un mélange et une suspension, suivis d'une redéposition; 2) Région du bassin central — Grossissement de l'est vers l'ouest du sédiment en équilibre de texture avec l'énergie hydraulique, en fonction d'une distance grandissante à parcourir sous l'influence des vents d'ouest et de sud-ouest prédominants; 3) Région du bassin est — Bassin profond dont la grosseur du sédiment va en diminuant vers le large à mesure que la profondeur augmente. Le sédiment des profondeurs est modifié par l'apport de quantités substantielles de matériaux de la grosseur du limon provenant de l'érosion de la ligne de rivage sur le côté nord du bassin central.

Les interrelations entre les paramètres indiquent une dépendance, sous le rapport de la texture, de la composition minéralogique, la relation entre la teneur en argile et la grosseur moyenne des grains ayant une importance particulière. Ceci a surtout de la valeur dans l'élaboration de modèles du comportement d'éléments géochimiques, tel le phosphore, associés à l'argile.

Accepted November 17, 1975 Accepté le 17 novembre 1975

THE sediment distribution and textural properties of the surficial sediments of Lake Erie have been studied as part of an ongoing program on the sedimentology and geochemistry of the Laurentian Great Lakes. The sedimentological characteristics of the sediments of Lakes Ontario and Huron have been previously reported (Thomas et al. 1972, 1973), Lake Superior (Thomas and Jaquet 1975), and the Big Bay Section of the Bay of Quinte, Lake Ontario (Damiani and Thomas 1974).

For many years, the sediments of Lake Erie have been a subject of considerable interest yet, other than the study of Lewis (1966), previous work has been confined to localized areas without consideration of the whole-lake processes. The first examination of bottom sediments in Lake Erie was carried out in Sandusky Bay by Moseley (1904). He discovered submerged stream valleys which, he concluded, were indicative of lower lake levels. Leverett and Taylor (1915) conducted a similar study in streams entering the Western basin of the lake. The first sediment distribution charts of the Central and Eastern basins were compiled by Fish (1929) and Pegrum (1929). The distribution of sand and gravel deposits in the Pelee Point region were described by Kindle (1933, 1936). A comprehensive study of the distribution of sediments throughout the lake was carried out by Lewis (1966), which provides the basis for the detailed sedimentological studies reported here.

Detailed nearshore studies on modern sediment deposits have been carried out and reported for the Ohio sector of the lake by Verber (1957), Hartley (1961a, 1961b) and for the Canadian shoreline from Pelee Point to the Niagara River by St. Jacques and Rukavina (1971 and 1973).

The geological and postglacial evolution of Lake Erie have been reviewed by Sly (1976) together with an overview of the watershed and lake bathymetry, and are not discussed further here. Additionally, both the geologic and postglacial history of the lake have been described in the context of the evolution of the Great Lakes basin by Sly and Thomas (1974).

The purpose of the present study is to relate the textural properties of the surface sediments of Lake Erie to the basin configuration and morphology and to achieve an understanding of sediment sorting processes as it relates to, or controls, sediment composition. The viability of this approach is demonstrated by the application of textural parameters in elucidating phosphorus forms and concentrations (Williams et al. 1976a) and by applying these results in the physical model for phosphorus regeneration (Lam and Jaquet 1976).

Methods

Sediment sampling, together with acoustic profiling, were carried out by the Canadian Department of the Environment vessel, CSS *Limnos*. Samples were collected at the intersections of a 10 km Universal Transverse Mercator (UTM) grid in the Central and Eastern basins of the lake (Fig. 1). In the Western basin, to obtain a closer sampling density, samples were taken at alternate intersections of a 5-km UTM grid (Fig. 1). The total sample network consisted of 275 sample locations from which 260 samples were obtained for textural and geochemical analyses. Sample recovery was poor or nil at the remaining sampling locations, due to the occurrence of coarse lag gravels and, less commonly, bedrock. Positioning was carried out by radar using a Decca 416 with variable range marker. The accuracy of the positioning is not known but is estimated as being better than ±500 m in mid-lake positions.

Continuous echosounding profiles were carried out between sampling stations using a Kelvin Hughes MS26B sounder, operating at 14.25 kHz in order to map the gross distribution of major sediment units (Thomas et al. 1972).

Shipek samples were collected at each location and temperature, Eh and pH were measured at a sediment depth of 1.5 cm and subsamples (75 cm^3) taken of

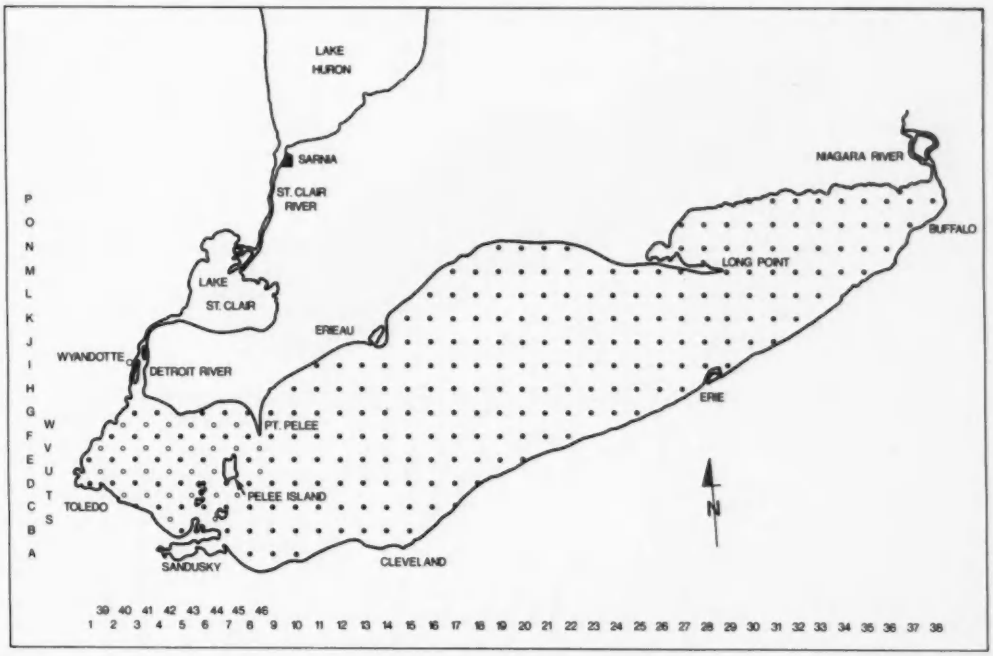

FIG. 1. Sample distribution plan.

the topmost 3 cm. Two subsamples were freeze-dried on board ship for later geochemical analysis, and two subsamples were stored at 4 C for later sedimentological analysis. Organic carbon, carbonate carbon, quartz, and particle-size determinations were carried out with the identical methods used for the sediments of lakes Ontario and Huron (Thomas et al. 1972, 1973). All sediment grain sizes are given phi units (phi $= -\log_2$[particle size diameter in mm]) and distribution statistics expressed as mean, standard deviation, skewness, and kurtosis (moment measures) were determined using the computer program SEDAN (Coakley and Beal 1972).

Sediment Distribution

The distribution of the surficial sediments of Lake Erie, based upon the interpretation of echograms with sample descriptions used as interpretive control is given in Fig. 2. The distribution is, in essence, the same as that prepared by Lewis (1966) with some modification of unit boundaries particularly in the Eastern basin. Five basic units are shown in Fig. 2, four of which are related to specific episodes in the glacial and postglacial evolution of the lake, modified by lake bathymetry. These units are identified and discussed as follows:

1) *Sands*: Offshore accumulations of sand are restricted in Lake Erie, with only one occurrence on the eastern margin of the Long Point–Erie sill (Lewis 1966). Major occurrences of sand and gravel occur in the Pelee Lorraine moraine south of Point Pelee but probably represent redistribution relict material (J. P. Coakley personal communication). Modern sands occur in the littoral transport zone and accumulate in major spit features at Long Point, Erie, Erieau, and in beach deposits.

2) *Glacial till and/or bedrock*: No distinction was made between till and bedrock in the interpretation of echograms. However, based on sample descriptions, till is the most predominant material with bedrock occurring in localized areas, in the island region north of Sandusky Bay (Lewis 1966) and in parts of the nearshore zone, particularly on the north shore of the Eastern basin (St. Jacques and Rukavina 1973). The till represents a nonhomogeneous mass of material composed of gravel, sand, and clay deposited during the northward ablation of the last ice mass. The till thus represents the oldest unconsolidated sediment within the lake basin, forming a veneer overlying bedrock. It crops out in the nearshore zone around the periphery of the lake and in the cross-lake sills (Pelee–Loraine, Long Point–Erie moraines). The till surface is exposed to high energy wave action and at all localities has a surface lag gravel.

3) *Glacio-lacustrine clay*: This represents the

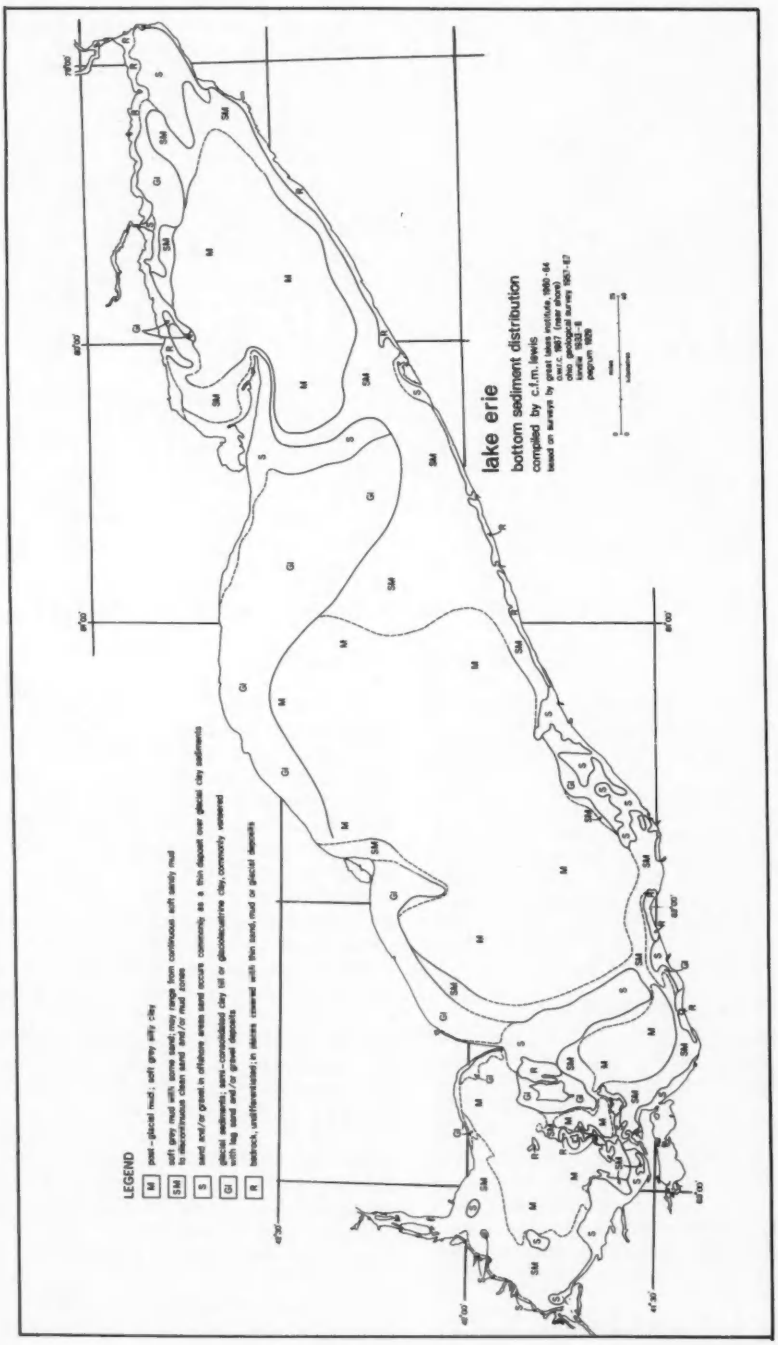
lake erie
bottom sediment distribution
compiled by c.f.m. lewis
based on surveys by great lakes institute 1950-54
usnhoc 1957 (near shore)
ohio geological survey 1957-67
usnhoc 1963-8
pagcon 1970
LEGEND
post-glacial mud; soft grey silty clay
soft grey mud with some sand, may range from continuous soft sandy mud to discontinuous clean sand and/or mud zones
sand and/or gravel in offshore areas sand occurs commonly as a thin deposit over glacial clay sediments
glacial sediments; semi-consolidated clay till or glaciolacustrine clay, commonly veneered with lag sand and/or gravel deposits
bedrock, unstratified and in places covered with thin sand, mud or glacial deposits
M
SM
S
GI
R

FIG. 2. Distribution of surficial sediments in Lake Erie.

FIG. 3. Basin distribution and nomenclature including cross-lake sills and inshore zone.

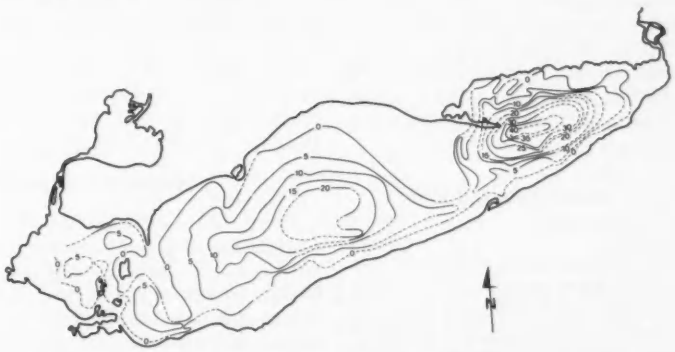

FIG. 4. Isopachyte map of recent sediment in Lake Erie (in meters).

product of subaqueous deposition in high water level lakes ponded by the ice mass to the north. The deposits are fine-grained, often varved with, and containing occasional ice-rafted erratics. The glacio-lacustrine sediments overlie the glacial tills and are exposed offshore of the till region and on the flanks of the cross-lake sills (Fig. 3). The glacio-lacustrine clay generally has a surface of lag sand 1 to 6 cm thick.

4) *Postglacial muds*: With the opening of the Niagara Gorge, the early drainage of Lake Erie through the Maumee River Valley to the Mississippi System changed to the Niagara Gorge, with northward drainage to the Lake Ontario basin. This resulted in the rapid lowering of lake levels and the formation of early Lake Erie. Since that time, differential isostatic uplift has resulted in rising water levels of approximately 33 m during the last 12,000 yr (Sly 1976). Continuing deposition during this time has resulted in the accumulation of the offshore deposits of fine-grained silty clays, which show a distribution markedly similar to lake bathymetry (Fig. 2 and Sly 1976). These muds (silty clay and clay) occur in four basins partially separated by the Pelee–Lorraine and Long Point–Erie sills. These basins are designated Western, Sandusky, Central, and Eastern basins, respectively. The recent sediments of the Central basin show a marked increase in grain size eastwards with the occurrence of sandy clays in the eastern and south eastern part of the basin (Fig. 2). The basin outlines and nomenclature defining the major morphometric features are summarized in Fig. 3.

The thickness of the basin muds shown in the isopachyte map (Fig. 4) after Lewis (1966) was determined by measurement of thickness from echosounder profiles. The zero isopachyte contour partitions the actively sedimenting basin regions of the lake from the nondepositional or even erosional regions occurring in the shallow water nearshore zone and on the cross-lake moraine sills. Maximum thickness of mud accumulation occurs in the Eastern basin with thicknesses in

FIG. 5. Distribution of Eh values at 1.5 cm sediment depth (volts).

excess of 40 m. Thicknesses in excess of 20 m occur in the Central basin with lower accumulation (less than 10 m) in the Western and Sandusky basins. Sedimentation rates for these basins are given by Kemp et al. (1976), together with budget for the fine-grained depositional sediments occurring in the basins.

Hydrogen Ion Concentration and Redox Potential

The hydrogen ion concentration (pH) and the redox potential (Eh) were measured at a sediment depth of 1.5 cm. The pH ranged from 6.9 to 8.0 with a mean of 7.4 and a standard deviation of 0.2. No general pattern was observed in the regional distribution of pH, though values in excess of 7.5 occurred in the sediments on the Long Point–Erie sill and in the embayment north of Long Point. The mean pH value of 7.4 is the same as that recorded for Lake Huron (Thomas et al. 1973) and only marginally lower than values recorded in the sediments of Lake Ontario (7.6 in the sands; 7.8 in the muds, Thomas et al. 1972).

Eh values showed considerable variation from sample to sample ranging from a low of −0.060 to 0.430 V. The mean value for all lake samples was 0.097 V with a standard deviation of 0.104 V. This compares with the mean values of 0.095 and 0.102 V for lakes Ontario and Huron, respectively. The distribution of Eh values in Lake Erie sediments is given in Fig. 5. The Eh shows no trends that are related to sediment type. Both the Western and Eastern basins have uniform Eh values ranging between 0 and 100 V. The western part of the Central basin has values ranging predominantly between 0.100 to 0.200 V. The highest values, in excess of 0.300 V, occur in association with the cross-lake sills. The lowest values −0.060 to 0.000 occur in the form of a plume, extending northward from the coast near Cleveland and curving eastward parallel to the south shore of the Central basin. This suggests a decrease in redox potential related to materials input from the Cayahuga River at Cleveland, Ohio.

Sediment Composition

Lewis (1966) carried out X-ray diffraction of the less than 8-μ grain-size fraction of Lake Erie sediments. He demonstrated a remarkable uniformity in mineralogy of this fraction which comprised quartz, feldspar, calcite, and dolomite. He additionally identified the clay mineralogy of the same fraction and demonstrated a dominance of illite which occurred in all samples with subsidiary chlorite. Kaolinite was also present in most of the samples examined. This clay mineral suite is identical to that observed in Lake Ontario (Thomas et al. 1972) and in Lake Huron (Thomas et al. 1973). No further X-ray diffraction work was conducted for this study and compositional variation was examined by the accurate determination of quartz plus feldspar, organic carbon, and inorganic carbon. No attempt was made to differentiate calcite from dolomite since this was previously undertaken by Lewis (1966).

QUARTZ

The gravimetric determination of total quartz plus feldspar was carried out after the method of Trostell and Wynne (1940). This method, based upon fusion with potassium pyrosulphate, preferentially removes the layered silicates (clays), organic and inorganic carbon, and sulphides, leaving a residue of quartz plus feldspar with resistant heavy minerals such as zircon. In this study no effort was made to differentiate the

TABLE 1. Summary table giving means and standard deviations of individual parameters.

	No. samples	Eh (V)		pH		Mean (φ)		Sand (%)		Silt (%)		Clay (%)		Org. C (%)		Inorg. C (%)		Quartz (%)		Depth (m)		
		$\bar{x}$	SD	$\bar{x}$	SD	$\bar{x}$	SD	$\bar{x}$	SD	$\bar{x}$	SD	$\bar{x}$	SD	$\bar{x}$	SD	$\bar{x}$	SD	$\bar{x}$	SD	$\bar{x}$	SD	CV
Total lake	227	0.097	0.104	7.37	0.21	6.43	2.57	24.6	33.9	32.2	16.9	43.3	24.2	2.18	1.03	1.00	0.88	49.2	17.1	20.2	9.9	.49
Inshore zone	83	0.108	0.111	7.47	0.24	3.64	2.17	62.4	29.4	20.5	17.8	17.4	14.5	1.24	0.84	1.17	1.02	66.8	12.30	16.6	6.3	.38
Total basins	144	0.090	0.099	7.31	0.18	8.04	0.80	2.9	4.0	38.9	12.0	58.2	13.8	2.72	0.69	0.89	0.77	39.1	9.70	22.3	10.9	.49
Western basin	30	0.048	0.033	7.31	0.15	7.83	0.67	3.3	4.2	42.6	8.9	54.0	10.2	2.93	0.56	1.45	0.99	37.0	5.97	9.8	1.7	.17
Sandusky basin	8	0.106	0.033	7.20	0.20	7.70	0.44	2.9	2.3	43.0	3.7	54.1	5.5	2.58	0.35	1.04	0.46	44.0	9.15	13.8	1.2	.09
Central basin	77	0.113	0.114	7.29	0.14	8.21	0.80	2.6	3.9	35.8	12.1	61.6	14.2	2.89	0.64	0.45	0.33	38.8	10.8	22.5	2.6	.12
Eastern basin	29	0.070	0.096	7.42	0.24	7.90	0.92	3.1	4.5	42.1	14.0	54.8	15.8	2.09	0.67	1.43	0.97	40.3	9.81	37.1	13.2	.35

FIG. 6. Distribution of Organic C in percent dry weight of sediment.

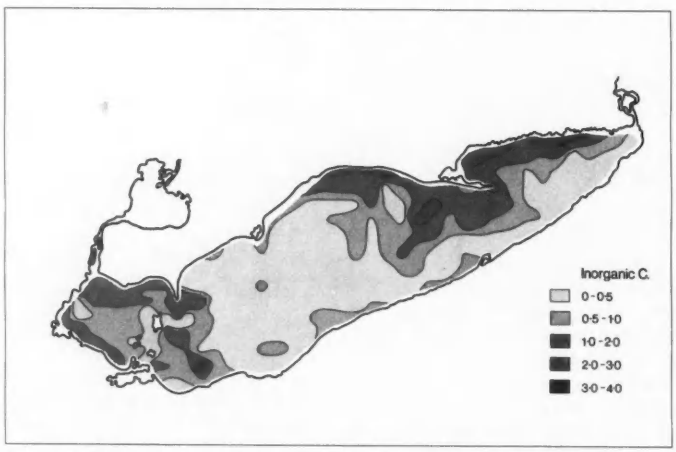

FIG. 7. Distribution of Inorganic C in percent dry weight of sediment.

feldspars as described by Thomas (1969a) for Lake Ontario. Reference to "quartz" in the following discussion is thus indicative of total quartz plus total feldspar expressed as percent dry weight of sediment.

Quartz in Lake Erie ranged from 25 to 90% with a mean of 49.2% and a standard deviation of 17.1%. No distribution map is given for the quartz as it shows a strong parallel distribution with both the sand-size fraction (Fig. 11) and mean grain size (Fig. 14). Higher values occur in the shallow-water zone around the lake periphery and in the sediments of the cross-lake sills, decreasing offshore into the depositional basins.

ORGANIC CARBON

Organic carbon in Lake Erie sediments ranges from 0.12 to 4.4%. Mean values together with standard deviations for individual basins, the inshore zone and total lake are given in Table 1. The mean value for Lake Erie (2.18%) is higher, though not significantly, than the mean values for lakes Ontario and Huron (1.98 and 1.63%, respectively).

The distribution of organic carbon is given in Fig. 6. The distribution is similar to that observed in the clay-size fraction (Fig. 13) and relates well to the basin configuration. Lowest values occur in the nearshore zone and the cross-lake sills, increasing offshore into the depositional basins. The organic carbon distribution in the Western basin is more interesting in that a concentration gradient can be observed out from the Detroit River suggesting input of organic-rich sediment from the Lake St. Clair–Detroit River system.

TABLE 2. Lower size limits of gravel, sand, silt, and clay fractions.

Fraction	Size Limits		
	mm	μ	ϕ units
Gravel	4		−2
Sand	0.063	63	4
Silt	0.004	4	8
Clay	<0.004	<4	>8

INORGANIC CARBON

Inorganic carbon in Lake Erie ranges from 0.01 to 3.7% with a mean of 1.0 and a standard deviation of 0.88. This mean is higher than the values of 0.43 and 0.52% recorded for the sediments of lakes Ontario and Huron, respectively. The mean values for inorganic carbon in the lake zones and individual basins are summarized in Table 1. These values show that inorganic C values are relatively high throughout the lake except in the Central basin which is significantly lower. The reason for this is uncertain at the present time but may well be, in part, related to a substantial variation in a detrital carbonate component.

The distribution of inorganic carbon is shown in Fig. 7. Values are low throughout the Central basin and in the southern part of the Eastern basin. The Western basin shows higher values (1.0–3.0%) close to the southern shore and values of 1.0–2.0% along the northern shore from the Detroit River to just east of Point Pelee. High values occur along the northern shore of the Central and Eastern basins, with a large area of high values south and southeast of Long Point. Additionally, high values can be observed throughout the embayment north of Long Point. Lewis (1966), showing the distributions of dolomite and calcite, observed a similar distribution of both components to that described above. This distribution, together with the occurrence of dolomite, suggests that the carbonate in the high carbonate areas is derived predominantly from shoreline erosion, with possible additional sources from subaqueous erosion of glacial deposits and carbonate input from the Detroit River derived from Lake St. Clair and Lake Huron. The significance of the carbonate content in terms of sediment texture will be discussed subsequently.

Sediment Texture

Folk and Ward (1957) interpreted the textural characteristics of river sediments in the Brazos River Bar as being the product of hydraulic mixing of two populations of primary sediment material composed of gravel and sand. (See Table 2 for definition of these terms.) On the basis of this mixing hypothesis, they derived a theoretical model based on the relationships between the particle-size measures of mean grain-size, standard deviation, skewness, and kurtosis. (See Table 3 for definition of these terms, and also Coakley and Beal 1972.) Spencer (1973) further evolved this concept and propounded an hypothesis that all detrital sediments could be defined in terms of grain-size parameters on the basis of a single population, or mixing of two or three populations, which he defined as occurring in the gravel-, sand-, and clay-size fractions. He suggested size ranges for these three populations

TABLE 3. Definition of moment measures.

Parameter	Moment	Value for normal distribution	Comments
Mean grain-size	First moment about zero	any	Large positive values = fine sediments
Standard deviation	Second moment about the mean	>0	Large values = poor sorting
Skewness	Standardized third moment about the mean	0.0	Positive values = distribution tail towards fine sizes. Negative values = tail towards coarse
Kurtosis	Standardized fourth moment about the mean	0.0	Large positive values = peaked or leptokurtic distribution. Large negative values = bimodal distribution

and stated that they showed a normal distribution if size is expressed in ϕ units.

In simplified form, the dependency of the grain-size characteristics on mixing of populations and their trends can be defined, provided that the nature of the populations involved is known. Thomas et al. (1972, 1973) in Great Lakes sediments defined the sand and clay populations as follows: sand, mean grain-size 1.5–3.5 ϕ, positively skewed and leptokurtic; clay, 7.0–9.0 ϕ, negatively skewed and leptokurtic. Commencing with a sand population and adding increasing concentrations of the clay population results in an inverse decrease in the sand, with grain-size parameter trends as follows:

Mean grain-size — grain-size decreases from 1.5 to 3.5 ϕ in a continuous sequence to 7.0–9.0 ϕ at which values only the clay population is represented.

Standard Deviation (Sorting) — sorting becomes progressively poorer (increasing standard deviation) then improves with decreasing sand until the standard deviation values of the clay population are reached.

Skewness — the positively skewed sand passes through normal distribution to attain the negative skewness of the clay population.

Kurtosis — the leptokurtic distribution is depressed through a platykurtic condition returning to the leptokurtic distribution of the clay population.

The significant difference between this model and the theoretical model of Spencer (1963) is the fact the two fundamental populations are asymmetric and do not conform to a normal distribution. Thomas et al. (1972) ascribed this lack of symmetry in the populations to the occurrence of a silt mode, which produces a tail in the distributions of both the sand and clay populations towards the silt. This results in the inverse skewness and leptokurtic character of the two populations. The viability of this hypothesis in the interpretation of sediment texture is evidenced by the growing number of studies on sedimentation in aqueous systems both lacustrine (Thomas et al. 1972, 1973; Vernet et al. 1972; Jaquet et al. 1975; Thomas and Jaquet 1975) and marine (Cronan 1969; Jones 1971). Deviation from the textural models may result from natural causes, for example change in the characteristics of the source materials, slumping, turbidity flows, or may well be indicative of incorrect sampling technique. On the basis of the working hypothesis outlined above, together with a recognition of potential causes of sample deviation, we will now discuss the observed trends in the textural properties of the sediments of Lake Erie.

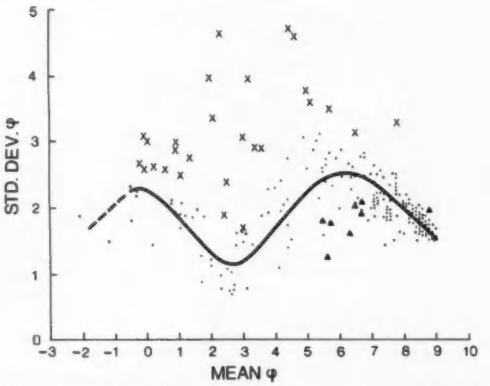

FIG. 8. Scatter diagram showing the relationship of mean grain-size to standard deviation (sorting), with theoretical trend line (See explanation in text).

RELATIONSHIPS BETWEEN SELECTED MEASURES

Mean grain-size vs. standard deviation — The relationship of mean grain-size and standard deviation is shown as a scatter diagram in Fig. 8. Low standard deviation can be observed in the regions 2 to 3, 8 to 9 ϕ and possibly at -2 ϕ. These regions of the trend curve are interpreted as being the end-member gravel, sand, and clay populations. Samples designated as ▲ on this same diagram have a high $CaCO_3$ content and fall into the silt fraction but, because of their low standard deviation, do not plot on the main trend. They represent the minor silt mode postulated by Thomas et al. (1972, 1973) and Jaquet et al. (1975). The approximated best fit trend line is sinusoidal as in Folk and Ward's (1957) model and shows the variations in both mean grain-size and sorting (standard deviation) when firstly, the sand and clay populations are mixed in a continuous series (3 to 8 ϕ, Fig. 8) and secondly,

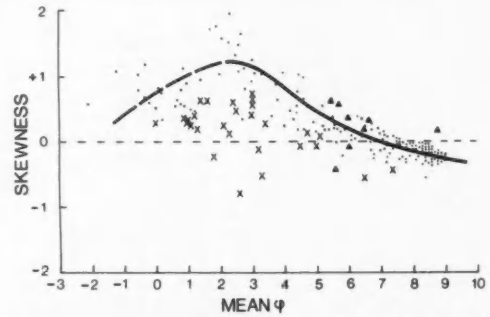

FIG. 9. Scatter diagram showing the relationship of mean grain-size to skewness with theoretical trend line (See explanation in text).

when a gravel population is mixed to the sand population (2 to −1 ϕ, Fig. 8). The samples designated × in this figure, showing marked deviation from the supposed natural trend line, were in all cases found to be nonrepresentative in that they were composed of thin lag gravels or sands (<3 cm) occurring over glacial clays in which both units had been included in the subsamples taken for analysis. On the basis of the scatter diagram, the size ranges and sorting characteristics (standard deviation) of the two primary populations forming the textural characteristics of Lake Erie sediments can be defined and compared with the suggested populations of Spencer (1963) and to other Great Lakes sediment populations as follows:

Sand Population	Range of Mean Grain Size (ϕ)	Range of SD (ϕ)
Lake Erie (this study)	1.5–4.5	0.6–1.6
Lake Huron (Thomas et al. 1973)	1.2–3.2	0.3–2.0
Lake Ontario (Thomas et al. 1972)	1.5–3.5	0.3–2.3
Lake Superior (Thomas and Jaquet 1975)	0.5–3.5	0.6–2.2
Spencer (1963)	1.5–4.0	0.4–1.0
Clay Population		
Lake Erie (this study)	7.0–9.0	1.4–2.6
Lake Huron (Thomas et al. 1973)	7.4–9.5	1.4–3.3
Lake Ontario (Thomas et al. 1972)	7.0–9.5	1.4–2.7
Lake Superior (Thomas and Jaquet 1975)	7.0–10.0	1.2–2.7
Spencer (1963)	7.0–9.0	2.0–3.0

In general, there is a remarkable agreement between the characteristics of the sand and clay populations for all of the studies listed above, with particularly good agreement between lakes Erie and Ontario. This implies a basic similarity in the source materials for the sediments of the Great Lakes. The further similarity with the theoretical populations of Spencer (1963) is indicative of a general applicability of these two populations in the interpretation of sediment textures.

Skewness vs. mean grain-size — Thomas et al. (1972, 1973), in the sediments of lakes Ontario and Huron, noted that the end-member sand and clay populations displayed a skewed distribution. As mentioned above, this again appears to be the case for the sediments of Lake Erie and is well-illustrated by the scatter diagram of skewness to mean grain-size shown in Fig. 9. The sand population (mean 1.5–4.5 ϕ) shows a positive skewness and the clay population (7.0–9.0 ϕ), a small negative skewness. Samples with a mean grain-size coarser than 1.5 ϕ tend to show a decrease in positive skewness values, suggesting a modification due to the addition of a coarser gravel population. A pure gravel population has not as yet been identified by the regional sediment studies in the Great Lakes and its characteristics remain unknown. Considering a two-population system, however, it is readily apparent that the dominant population can be isolated by the sign of the skewness in addition to the mean grain-size of the sample. If the anomalous samples are singled out, as in Fig. 9, a sinusoidal-like trend line can be drawn, which expresses the ideal variation of skewness with mean. Its shape is analogous to that postulated by Folk and Wards' (1957) model, with the difference that, in Lake Erie, end-member populations are skewed.

Skewness vs. kurtosis — It was demonstrated by Thomas et al. (1972, 1973) that the sand and clay populations in the Great Lakes deviated from a normal distribution not only in terms of skewness, but also in kurtosis (peakedness). They showed that both populations were leptokurtic (peaked curve) with stronger leptokurtism in the sand. Similar characteristics are evident for the populations in Lake Erie as discussed below. In terms of grain-size characteristics, it can thus be stated that the Lake Erie sand and clay populations are identical to those observed in lakes Superior, Huron, and Ontario, in that the sands are strongly leptokurtic with positive skewness and the clays weakly leptokurtic with a negative skewness. Mixing of the two populations results in changing values of both parameters as can be seen in the scatter diagram given in Fig. 10, showing the relationship of kurtosis to skewness. This diagram has been subdivided in four zones, A, B, C, and D, following Thomas et al. (1972). Zone A shows positive skewness with a leptokurtic distribution and incorporates the sand and gravel populations. Zone D shows negative skewness and low positive kurtosis (leptokurtic) and incorporates the clay population. Zones B and C are both platykurtic (flattened distribution curve due to biomodality) and represent mixed sand and clay sediments with the dominant population indicated by the sign of the skewness. Zone B is positively skewed with dominant sand, whereas Zone C is negatively skewed with a predominant clay population. The trend line denoted in Fig. 10 passing from Zone A through B and C to D approximates the changes in both kurtosis and skewness in a continuously mixed series from sand through to clay. This line is interpreted as representing declining energy from a high-energy Zone A to a low-energy Zone D.

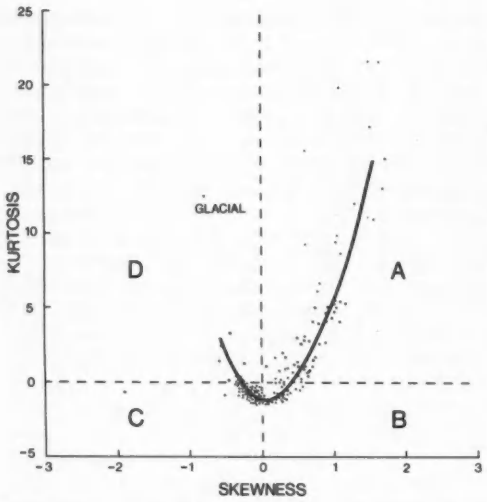

FIG. 10. Scatter diagram showing the relationship of skewness to kurtosis with theoretical trend line.

AREAL DISTRIBUTION OF SOME TEXTURAL PARAMETERS

The lakewide distribution of sand, silt, and clay percentages as well as mean grain-size are given in Fig. 11–14. As these parameters are strongly correlated (see Table 4) and provide similar information, only the map of mean grain-size distribution will be discussed in detail (Fig. 14).

As previously stated, the statistical grain-size distribution parameters, including mean grain-size, are related to the hydraulic mixing of a sand and clay population. Mean grain-size is thus a measure of the degree of mixing in a continuous series between the end-member populations. The distribution of mean grain-size thus provides some information on the degree of mixing as it relates to depositional energy and to a lesser extent provides information on the availability of source material.

The distribution of mean grain-size is shown in Fig. 14. Coarse sediments occur around the periphery of the lake, particularly along the northern shore of the Central basin between Point Pelee and Long Point. Further extensive areas of coarse sediment occur on the Pelee–Lorraine sill and on the Long Point–Erie sill. These coarser deposits occur in nondepositional areas as defined previously (Fig. 2) and demonstrate that the lag deposits of sand and fine gravel, overlying glacial clays and occurring under nondepositional and erosional conditions, are in dynamic equilibrium with the applied environmental hydraulic energy. This is discussed subsequently. The finer sediments (Fig. 14) occur in the depositional basins of the lake with a mean grain-size predominantly in the 8–9 ϕ size range. Mean grain-size in the Eastern basin shows a relationship with bathymetry as shown by a correlation coefficient of 0.784 between mean grain-size and water depth (Table 4). Such a strong relationship is not observed in the remaining basins. The Central basin exhibits a marked trend of increasing grain-size from west to east along the main axis of the lake. This is believed to be due to increasing wave energy with increasing fetch relative to the prevailing westerly and southwesterly winds. Mean grain-size distribution in the Western basin shows a plumelike pattern implying a major source of fine-grained sediment from the Detroit River.

Some of the high-silt zones visible on the map of silt distribution (Fig. 12), such as in the embayment north of Long Point, could be related to calcium carbonate precipitation, as indicated by the higher inorganic carbon content of these sediments.

ENERGY ZONES AND DEPOSITIONAL PROCESSES

The distribution of the samples into four zones, A, B, C, and D, on the diagram skewness–kurtosis (Fig. 10) is plotted in Fig. 15. This figure, if interpreted as an energy distribution map for the sediments of Lake Erie, shows characteristics which adequately explain variations in sorting observed in the lake sediments. Firstly, the high-energy Zone A occurs in the nearshore areas particularly on the northern shore of the Central basin and the eastern shore of the Eastern basin. (These shorelines being the most subjected to the prevailing westerly and southwesterly winds.) Zone A, in addition, occurs prominently on the Pelee–Lorraine and Long Point–Erie sills. The four main basins of the lake consist predominantly of Zone C with only small areas of Zone D. This is in contrast to lakes Ontario and Huron (Thomas et al. 1972, 1973) where the basin deposits were predominantly in low-energy Zone D. The inference from the incidence of Zone C in the Lake Erie basins suggest that this lake is of an overall higher energy nature than the other Great Lakes, presumably because of its lesser depth. This is certainly valid for the Western, Central, and Sandusky basins but appears to be anomalous with respect to the deepwater Eastern basin. This basin, on the basis of observations in Lake Ontario (Thomas et al. 1972) should be in Zone D. However, it is believed that direct input of coarse clay and fine silt, derived from shoreline

FIG. 11. Distribution of percent sand-size fraction in the surficial sediment of Lake Erie.

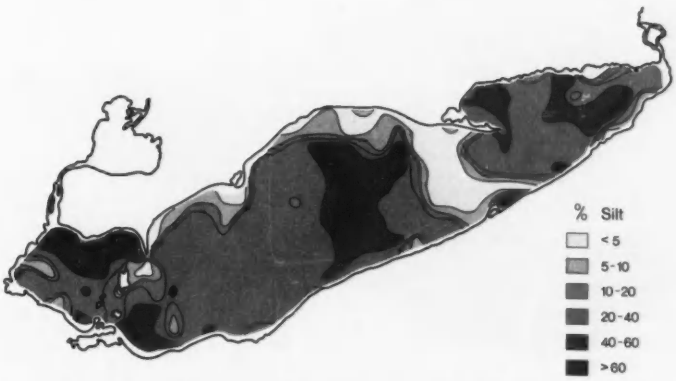

FIG. 12. Distribution of percent silt-size fraction in the surficial sediments of Lake Erie.

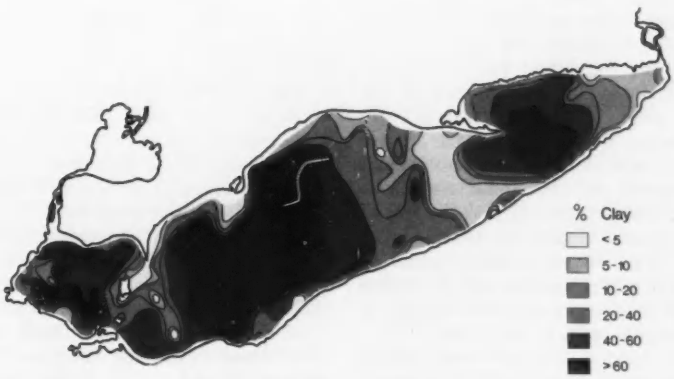

FIG. 13. Distribution of percent clay-size fraction in the surficial sediments of Lake Erie.

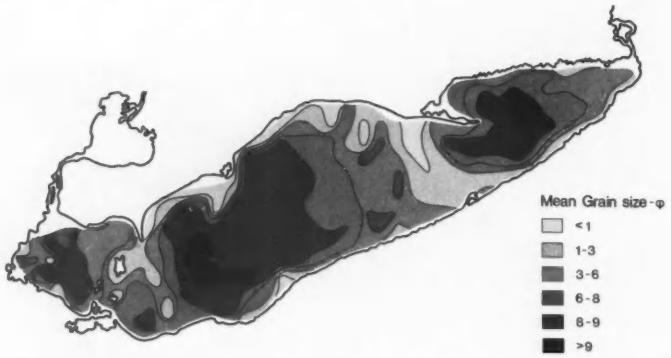

FIG. 14. Distribution of mean grain-size in phi in the surficial sediments of Lake Erie.

FIG. 15. Distribution of energy Zones A, B, C, and D in Lake Erie as derived from the relationship of skewness to kurtosis given in Fig. 10.

erosion, is occurring in the Eastern basin from Long Point. (Long Point is accreting eastwards at a rate of 7 m per annum, Lewis 1966.) This input has a modifying effect, particularly on kurtosis, in maintaining these sediments in the Zone C energy classification. The Central basin shows an interesting west to east trend of increasing energy related to increasing fetch under prevailing wind conditions. A similar trend was observed in the shallow water Big Bay Section of the Bay of Quinte, Lake Ontario by Damiani and Thomas (1974). The Western basin sediments fall predominantly into Zones B and C, which relate well with higher energy applied to fundamentally fine-sediment materials. It is interesting to note that, despite the fact that in mean grain-size, percent sand, silt, and clay, Lake Erie shows very close agreement with the sediment of Lake Ontario, the resolution provided by the skewness to kurtosis relationship suggests very positively that Lake Erie represents a higher energy situation, presumably derived from its shallow-water nature.

The hydrodynamic trend line between A and D zones has been plotted on the ternary sand–silt–clay diagram and is shown in Fig. 16. Three fields may be defined in this diagram: A) "Normal" samples, falling close to the hydrodynamic trend line, B) Till and till-contaminated samples, showing a tendency towards bimodal mixing of clay and sand populations with a dearth of material in the intermediate silt size (Thomas et al. 1972), C) Carbonate-rich samples, showing the presence of a true silt mode. The same categories have been found in surface samples of Lake Geneva (Jaquet et al. 1975). Similar distributions for normal samples have been observed in ternary diagrams by Pelletier (1969, 1973, 1974).

TABLE 4. Summary table of correlation coefficients. (* Significant at .05 level, ** Significant at .01 level.)

A. Relationships with the Sand-Size Fraction

	# Samples	Quartz	Depth	Silt	Clay	Mean grain-size	Inorg. C
Total	227	.826**	−.259**	−.739**	−.881**	−.951**	−.130
Inshore Zone	83	.615**	.002	−.933**	−.883**	−.900**	−.064
Total basin	144	.590**	−.251*	.327**	−.572**	−.573**	.355**
Western basin	30	.573**	−.354	.087	−.484**	−.648**	.178
Sandusky basin	8	.510	−.455	.642	−.855**	−.958**	.294
Central basin	77	.620**	−.464**	.425**	−.635**	−.518**	.490**
Eastern basin	29	.663**	−.630**	.248	−.508**	−.618**	.549**

B. Relationships with the Silt-Size Fraction

	# Samples	Quartz	Depth	Clay	Mean grain-size	Inorg. C
Total	227	−.368**	.044	.334**	.571**	.098
Inshore Zone	83	−.514**	.002	.665**	.794**	.062
Total basin	144	.726**	−.210	−.962**	−.858**	.383**
Western basin	30	.685**	.009	−.914**	−.749**	.497**
Sandusky basin	8	.545	−.235	−.947**	−.565	.264
Central basin	77	.833**	−.150	−.969**	−.855**	.184
Eastern basin	29	.664**	−.630**	−.961**	−.905**	.344

C. Relationships with the Clay-Size Fraction

	# Samples	Quartz	Depth	Mean grain-size	Inorg. C	Org. C	Eh
Total	227	−.898**	.326**	.931**	−.245**	.797**	−.106
Inshore Zone	83	−.622**	−.037	.855**	.070	.575**	−.317**
Total basin	144	−.801**	.255*	.910**	−.435**	.550**	.101
Western basin	30	−.835**	.136	.922**	−.510**	.118	−.141
Sandusky basin	8	−.584	.349	.785*	−.302	.704	−.197
Central basin	77	−.880**	.254*	.871**	−.290*	.748**	.096
Eastern basin	29	−.781**	.741**	.982**	−.464**	.419**	−.081

D. Relationships with Mean Grain-Size Fraction

	# Samples	Quartz	Depth	Sand	Inorg. C	Org. C	Eh
Total	227	−.827**	.298**	−.951**	−.184*	.751**	−.174
Inshore Zone	83	−.443**	.012	−.900**	.035	.462**	−.350**
Total basin	144	−.765**	.294*	−.573**	−.409**	.504**	.043
Western basin	30	−.859**	.144	−.648**	−.459*	.249	−.174
Sandusky basin	8	−.568	.397	−.958**	−.229	.549	−.163
Central basin	77	−.794**	.324**	−.518**	−.297*	.641**	.068
Eastern basin	29	−.813**	.784**	−.618**	−.460**	.461**	−.186

It was stated by Thomas et al. (1973) on the sediments of Lake Huron, that "since the energy is derived predominantly from wind stress, then initial energy will be a function of wind strength and direction related to lake surface area and fetch. It is likely, therefore, that all lake sediment systems will respond as a direct function of total initial energy and depth." This implies that in lake systems, wave orbital motion is the prime energy form resulting in sediment sorting, and maybe transport. This hypothesis is at present being tested by one of us (J.M.J.), using a wave hindcasting method and computing bottom orbital velocities at daily intervals for the year 1970.

Lewis (1966), basing his observations primarily on water depth, concluded that no deposition was occurring in the Western basin, with little or no sedimentation in the Central basin compensated by exceedingly rapid sedimentation in the deep-water Eastern basin. This would imply that the export of sediment from Western to Central and Central to Eastern basins is equivalent to the input sediment loadings to these two major westerly basins. The pollen observations of Kemp et al.

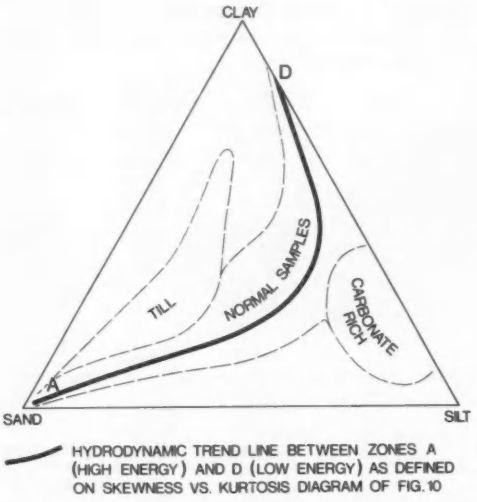

FIG. 16. Tenary diagram based on percent sand, silt, and clay, showing division into compositional zones of Lake Erie sediments.

(1976) demonstrate that sedimentation is occurring rapidly in all of the Lake Erie basins. A calculation of the loadings relative to the basin sedimentation for the Western and Central basins from the data presented by Kemp et al. (1976) shows quite clearly that input is considerably in excess of output, which by necessity results in net accretion in the basins. With particular reference to the Western basin, it is anomalous to find fine-grained materials occurring under the observed shallow-water conditions. However, bearing in mind the imbalance in the sediment input to export budget with a paucity of coarse bedload materials from the Detroit and Maumee rivers, then the sediments of the Western basin represent a redistribution of the large quantity of fine-grained sediment deposited from the suspended loads of these two rivers. The above discussion would suggest that the grain-size characteristics of the surface sediments of the Western basin are not in equilibrium with the available hydraulic energy. It is believed th ، the excess wave orbital energy in the basin, relative to the fine-grained bottom sediment, is utilized in resuspension and mixing processes. This mechanism would appear to be supported by the nutrient and trace metal core data presented by Walters et al. (1974), Kemp et al. (1976), and Williams et al. (1976b) in which sediment mixing appears to occur to sediment depths of 14–20 cm in the Western basin. It is possible, however, that this mixing may be due to bioturbation; yet the sediments of the Western basin suggest that physical mixing is highly significant and worthy of in-depth study.

Interrelationships Between Parameters and Discussion

The interrelationships between parameters is discussed in terms of the correlations shown by selected variables with the sand, silt, and clay-size materials. Correlation coefficients are summarized in Table 4.

RELATIONSHIPS WITH THE SAND-SIZE FRACTION (TABLE 4A)

Quartz shows a significant correlation with sand throughout the lake, similar to that observed in lakes Ontario and Huron. In general, the sand fraction is not related to depth though, in the basin sediments, there is a characteristic inverse relationship, particularly in the Eastern basin, demonstrating an offshore decrease in sand content as observed in lakes Ontario and Huron (Thomas et al. 1972, 1973). The total lake samples show a strong inverse relationship of clay and mean grain-size to the sand-size fraction. The relationship to clay is essentially a function of the inverse relationship expected in percentage components (in this case, sand, silt, and clay) summing to 100% (Chave and Mackenzie 1961). The relationship to mean grain-size is significant in that it confirms the variation to be expected in mean grain-size relative to the variable mixing of essentially a two population system as described earlier.

The relationship to silt is interesting in that an inverse silt to sand relationship is observed for the total samples and inshore zone, whereas poorer positive relationships can be observed in the basin sediments. This implies the occurrence of a silt population related inversely to sand in the high-energy nearshore zone, which has not been previously described as such in the sediment texture. This is discussed subsequently. Other than in the Central and Eastern basins, no relationship can be observed between inorganic C and the sand-size fraction. In these two basins, however, a significant correlation can be observed, indicating a covariance of a sand-size carbonate with the sand fraction in the basin sediments. As discussed previously, much of the carbonate in the Lake Erie sediments is believed to be of detrital origin from shoreline erosion. The sand-size component of this detrital material has thus been subject to sorting in these areas of sediment accumulation. No relationship with sand occurs in the inshore zone, even though this zone contains the highest values for inorganic carbon,

suggesting that the occurrence of carbonate in this zone is random and not related to a specific size fraction (see Table 4A, B, and C) and hence has not been subjected to redistribution relative to sorting processes.

Relationships with the Silt-Size Fraction (Table 4B)

Quartz shows a significant positive correlation with silt in the basin sediments with an inverse relationship in the inshore zone. This suggests that the silt-size materials consist predominantly of quartz and feldspar in Lake Erie basin sediments. This was not observed for lakes Ontario or Huron and is believed to be due to the proportionally high input of sediment derived from shoreline erosion, particularly from the north shore of the lake (Kemp et al. 1976). This reflects a high-proportion of finely-ground rock flour, originally derived from ice scouring, contained in the glacial tills of the shoreline bluffs. In many respects, it bears similarity to the B-type basin deposits of northern Lake Huron described by Thomas et al. (1973). The textural characteristics of this quartz-type, silt-size material have not yet been determined but, in certain samples, a fairly persistent mode has been observed at 4.5 ϕ in the coarse-silt range which may reflect this population. The major effect on the textural properties has been a modification of the skewness and kurtosis of the sand population. The inverse relationship of silt to sand (Table 4A) and of quartz to silt in the inshore zone (Table 4B) suggests that in the clay-deficient, high-energy regions, this silt mode has been acting as a discrete population. This is further corroborated by the inverse and very significant relationship of clay to silt in the basin sediments. It can thus be concluded that the silt fraction in Lake Erie sediments is behaving not only as a modifying mode on the textural characteristics of the end-member primary sand and clay populations, but also may be playing a discrete role as a coarse silt population. Compositionally, it may be incorporated with the sand since it is composed primarily of finer-grained, detrital materials derived from shoreline erosion.

The relationship of silt to inorganic carbon observed in Lakes Ontario and Huron (Thomas et al. 1972, 1973) is not convincingly demonstrated in Table 4B. The relationships observed in the Total, Western, and Eastern basin samples (Table 4B) indicate that a carbonate silt mode may be present. Indeed, the ternary diagram given in Fig. 16 provides ample evidence for its occurrence, though the degree of the relationship, particularly in the Central basin, as observed in Table 4, suggests masking by the quartz–silt relationship discussed above.

Relationships with the Clay-Size Fraction (Table 4C)

The clay-size fraction is behaving well as a relatively unmodified primary end-member population, as evidenced by the inverse and consistent relationship of clay to quartz (Table 4C). The behavior of the clay-size fraction as a primary population with its associated impact on mean grain size is further well illustrated by the clay to mean grain-size relationships given in Table 4C. This latter relationship is particularly important since it demonstrates a direct relationship of texture and mineralogical composition (see Thomas 1969b). Williams et al. (1976a) have demonstrated the relationship of nonapatite inorganic phosphorus (NAIP) to clay and associated iron. Since clay content can be predicted in these sediments, and indeed in the other Great Lakes' sediments examined to date on the basis of mean grain-size, then NAIP can be computed indirectly from mean grain-size. This factor has been utilized effectively in modelling the physical regeneration of phosphorus in Lake Erie by Lam and Jaquet (1976).

It can further be observed in Table 4C that, other than in the Western basin, a good relationship exists between organic carbon and clay. The significance of this factor to the modelling of organic carbon associated trace metals and nutrients can be inferred by analogy to the clay–phosphorus modelling noted above. The breakdown in the organic carbon to clay relationship in the Western basin (Table 4C) is believed to be due to direct organic carbon loading from the Detroit River as discussed previously.

No direct relationship can be observed between clay and Eh. However, correlation coefficients show that a good relationship of Eh to organic carbon occurs only in the Eastern basin (−0.703). This agrees with previously noted relationships in Lake Ontario and Lake Huron (Thomas et al. 1972, 1973). The Eh distribution given in Fig. 5 demonstrated the strictly local impact of the Cayahuga River on the redox potential of the sediments of the Central basin. Such effects would negate a functional organic carbon–Eh relationship, and it is this type of influence from the major tributaries which is believed to mask the relationship in both the Central and Western basins of the lake.

Summary and Conclusions

The texture of the surficial sediments of Lake Erie show similar characteristics to those de-

scribed for lakes Ontario, Huron, and Superior. These characteristics, when defined by moment measures, demonstrate spatial trends which can be interpreted on the premise of proportional mixing of a two-population system. These populations are composed of sand and clay, with basic characteristics inherited from the source materials originating predominantly from shoreline erosion with subsidiary loading from the tributary watersheds. Lake Erie sediments further provide evidence of a silt mode, which appears to behave as a discrete population. On the basis of statistical relationships, this silt is composed predominantly of finely comminuted quartz and is derived from shoreline erosion. A further silt mode, consisting of carbonate, has been identified in agreement with the findings of Thomas et al. (1972 and 1973) on the sediments of lakes Ontario and Huron. The statistical relationship of silt to carbonate, however, is not clear due to the occurrence and masking effect of the quartz-silt mode on the correlation noted above. The trends observed in mean grain-size, skewness, and kurtosis relative to proportional population mixing have been used to infer the hydraulic energy available at the sediment–water interface and its effect on sediment sorting. On the basis of this sorting, Lake Erie can be subdivided into three distinct regions in which differing responses to environmental energy can be observed:

1) *Western Basin Region* — Fine-grained sediment predominates under shallow-water conditions. High loadings of fine-grained sediment are derived from the Detroit and Maumee rivers with a deficit of coarser materials from river bedload and shoreline erosion. Input loadings are in excess of sediment export to the Central basin region, resulting in net accretion of sediment. The environmental energy from wave action, based on the shallow-water conditions, is believed to be in excess of that required to account for the observed sediment characteristics. This results in reworking, by mixing and resuspension, of the fine-grained sediment materials of the Western basin. The condition may thus be described as being in textural disequilibrium.

2) *Central Basin Region* — This region is again of a shallow-water nature. However, sediment sorting, as indicated by mean grain-size, skewness, and kurtosis, shows a distinct trend for a west to east coarsening of sediment due to increasing fetch and energy associated with prevailing westerly and southwesterly winds. Materials available from shoreline erosion consist of all grain sizes and the sediments are in equilibrium with the available hydraulic energy.

3) *Eastern Basin Region* — This is a deepwater basin, showing characteristic depth control of sediment texture as observed in previous studies on lakes Ontario and Huron. The sediments in this region show an offshore decrease in mean grain-size, reflecting decreasing energy with increasing water depth. On the basis of energy distribution derived from the relationship of skewness to kurtosis, the deepwater Eastern basin indicates higher energy than would be anticipated from the studies on lakes Ontario and Huron. This variation is due to the influx of silt-size material (silt population discussed above) derived from easterly transportation from Long Point directly into the deep waters of the Eastern basin. As noted above, this silt-size detrital material is derived from the large-scale erosion of the north shore of the Central basin region.

The interrelationships between parameters confirms that texture, in responding to the hydraulic sorting of two predominant populations, is essentially a control on gross mineralogic composition. This is particularly true of the clay percentage which shows direct relationships with the textural properties. On this basis, mean grain-size can be utilized to predict clay content and hence clay-associated geochemical elements. This relationship has been demonstrated with regard to phosphorus by Williams et al. (1976a) and successfully applied by Lam and Jaquet (1976) in modelling the physical regeneration, by sediment resuspension, of total phosphorus in Lake Erie.

Acknowledgments

The authors would like to thank their many colleagues at C.C.I.W. for their helpful discussion during the period of this study. Thanks are expressed particularly to T. W. Morton for analytical support, the crew of the C.S.S. *Limnos* and staff of the sedimentology laboratory for their enthusiasm and support. Thanks are also expressed to Marilyn Stapleton for typing the manuscript.

CHAVE, K. E., AND F. T. MACKENZIE. 1961. A statistical technique applied to the geochemistry of pelagic muds. J. Geol. 69: 572–582.

COAKLEY, J. P., AND G. S. BEAL. 1972. SEDAN — A computer program for sediment particle size analysis. Can. Inland Waters Dir. Rep. Ser. 20: 33 p.

CRONAN, D. S. 1969. Recent sedimentation in the central northeastern Irish Sea. Inst. Geol. Sci. London, Rep. 69: 10 p.

DAMIANI, V., AND R. L. THOMAS. 1974. The surficial sediments of the Big Bay section of the Bay of Quinte, Lake Ontario. Can. J. Earth Sci. 11: 1562–1576.

FISH, C. J. 1929. Preliminary report on the cooperative survey of Lake Erie. Buffalo Soc. Nat. Sci. Bull. 14: 7–270.

FOLK, R. L., AND W. C. WARD. 1957. Brazos River Bar: A study in the significance of grain size parameters. J. Sediment. Petrol. 27: 3–26.

HARTLEY, R. P. 1961a. Bottom deposits in Ohio waters of central Lake Erie. Ohio Div. Shore Erosion, Dep. Nat. Resour. Tech. Rep. 6: 14 p.

1961b. Bottom sediments in the island area of Lake Erie. Ohio Div. Shore Erosion, Dep. Nat. Resour. Tech. Rep. 9: 22 p.

JAQUET, J.-M., J.-P. VERNET, AND R. L. THOMAS. 1975. Etude granulométrique des sédiments du lac Léman. Abstr. Int. Sediment. Congr., Nice (France). 11: 8 p.

JONES, A. S. G. 1971. A textural study of marine sediments in a portion of Cardigan Bay (Wales). J. Sediment. Petrol. 41: 505–516.

KEMP, A. L. W., R. L. THOMAS, C. I. DELL, AND J.-M. JAQUET. 1976. Cultural impact on the geochemistry of sediments in Lake Erie. J. Fish. Res. Board Can. 33: 440–462.

KINDLE, E. M. 1933. Erosion and sedimentation at Point Pelee. Ont. Dep. Mines, Ann. Rep. 42: 1–29.

1936. Geology of Pelee and adjacent islands. Ont. Dep. Mines, Ann. Rep. 45: 75–116.

LAM, D. C., AND J.-M. JAQUET. 1976. Computation of physical transport and regeneration of phosphorus in Lake Erie, fall 1970. J. Fish. Res. Board Can. 33: 550–563.

LEVERETT, F., AND F. B. TAYLOR. 1915. The pleistocene of Indiana and Michigan and the history of the Great Lakes. U.S. Geol. Surv. Monogr. 53: 529 p.

LEWIS, C. F. M. 1966. Sedimentation studies of unconsolidated deposits in the Lake Erie Basin. Ph.D. Thesis. Univ. Toronto, Toronto, Ont. 134 p.

MOSELEY, H. L. 1904. Formation of Sandusky Bay and Cedar Point. Ohio State Acad. Sci. 4: 179–238.

PEGRUM, R. H. 1929. Topography of the Lake Erie Basin. Buffalo Soc. Nat. Sci. Bull. 14: 17–24.

PELLETIER, B. R. 1969. Submarine physiography, bottom sediments and model of sediment transport, p. 100–135. *In* Earth Sci. Symp. Hudson Bay. Geol. Surv. Can. Pap. 68-53.

1973. A re-examination of the use of silt/clay ratios as indicator of sedimentary environments: A study for students. Marit. Sediments. 9: 1–12.

1974. Sedimentary textures and relative entropy and their relationship to the hydrodynamic environment, Bay of Fundy system, p. 77–96. *In* Offshore Geol. East. Canada, Vol. 1. Geol. Surv. Can. Pap. 74-30.

RUKAVINA, N., AND D. A. ST. JACQUES. 1971. Lake Erie nearshore sediments — Fort Erie to Mohawk Point. Proc. Conf. Great Lakes Res. 14: 387–393.

ST. JACQUES, D. A., AND N. RUKAVINA. 1973. Lake Erie nearshore sediments — Mohawk Point to Port Burwell, Ontario. Proc. Conf. Great Lakes Res. 16: 454–467.

SLY, P. G. 1976. Lake Erie and its basin. J. Fish. Res. Board Can. 33: 355–370.

SLY, P. G., AND R. L. THOMAS. 1974. Review of geological research as it relates to an understanding of Great Lakes limnology. J. Fish. Res. Board Can. 31: 795–825.

SPENCER, D. W. 1963. The interpretation of grain-size distribution curves of clastic sediments. J. Sediment. Petrol. 33: 180–190.

THOMAS, R. L. 1969a. The qualitative distribution of feldspars in surficial bottom sediments from Lake Ontario. Proc. Conf. Great Lakes Res. 12: 364–379.

1969b. A note on the relationship of grain-size, clay content, quartz and organic carbon in some Lake Erie and Lake Ontario sediments. J. Sediment. Petrol. 39: 803–809.

THOMAS, R. L., A. L. W. KEMP, AND C. F. M. LEWIS. 1972. Distribution, composition and characteristics of the surficial sediments of Lake Ontario. J. Sediment. Petrol. 42: 66–84.

1973. The surficial sediments of Lake Huron. Can. J. Earth Sci. 10: 226–271.

THOMAS, R. L., AND J.-M. JAQUET. 1975. The surficial sediments of Lake Superior. Abstr. Int. Sediment. Congr., Nice (France). 11: 7 p.

TROSTELL, L. J., AND D. J. WYNNE. 1940. Determination of quartz (free silica) in refractory clays. J. Am. Ceram. Soc. 23: 18–22.

VERBER, J. L. 1957. Bottom deposits of western Lake Erie. Ohio Div. Shore Erosion. Dep. Nat. Resour. Tech. Rep. 4: 4 p.

VERNET, J.-P., R. L. THOMAS, J.-M. JAQUET, AND R. FRIEDLI. 1972. Texture of the sediments of the Petit-Lac (western Lake Geneva) Ecologae Geol. Helv. 65: 591–610.

WALTERS, L. J., T. L. KOVACIK, AND C. E. HERDENDORF. 1974. Mercury occurrence in sediment cores from western Lake Erie. Ohio J. Sci. 74: 1–19.

WILLIAMS, J. D. H., J.-M. JAQUET, AND R. L. THOMAS. 1976a. Forms of phosphorus in the surficial sediments of Lake Erie. J. Fish. Res. Board Can. 33: 413–429.

WILLIAMS, J. D. H., T. P. MURPHY, AND T. MAYER. 1976b. Rates of accumulation of phosphorus forms in Lake Erie sediments. J. Fish. Res. Board Can. 33: 430–439.

Mercury in the Surficial Sediments of Lake Erie

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THOMAS, R. L., AND J.-M. JAQUET. 1976. Mercury in the surficial sediments of Lake Erie. J. Fish. Res. Board Can. 33: 404–412.

In 259 samples of the topmost 3 cm of sediment taken from Lake Erie in 1971, total mercury values ranged from 8 to 2929 ppb ($\bar{x}$ 582 ± 555 ppb SD). The highest levels were in the Western basin adjacent to the Detroit River, the rest of the lake showing increasing concentration from the coarser sediments of the Inshore Zone and cross-lake moraine sills out into the basin muds. A quartz correction was applied to compensate for texture and dilution by an inert constituent. The distribution of quartz-corrected mercury confirmed that the Detroit River is the predominant source of industrial mercury to the Western basin with subsequent transport across the Central basin to final sink in the Eastern basin. Further possible inputs were indicated from Erie, Pennsylvania and Buffalo, New York. Analysis of suspended sediment in the Detroit River showed no change in concentration between 1970 and 1974 with a mean 1974 level of 1060 ppb. Mercury loading was estimated at 7.2 m.t./yr to the sediments of the Western basin. A relationship was observed between total mercury and organic carbon, showing two distinct regressions: 'A' with lower organic-bound mercury, is taken to represent the levels related to industrial and natural loadings existing prior to the introduction of mercury cell chlor-alkali plants; 'B' applicable to highly contaminated sediments, is believed to be indicative of loadings derived from the chlor-alkali industry.

THOMAS, R. L., AND J.-M. JAQUET. 1976. Mercury in the surficial sediments of Lake Erie. J. Fish. Res. Board Can. 33: 404–412.

Dans 259 échantillons des 3 cm supérieurs de sédiment prélevé dans le lac Erié en 1971, les valeurs de mercure total varient de 8 à 2929 pp10^9 ($\bar{x}$ 582 ± 555 pp10^9 SD). Les niveaux les plus élevés se trouvent dans le bassin ouest avoisinant la rivière Détroit, le reste du lac montrant des concentrations augmentant des plus gros sédiments de la zone littorale et les filons-couches de moraine à travers le lac jusqu'aux vases du bassin. Nous avons appliqué une correction quartz afin de compenser pour la texture et la dilution par un constituant inerte. La répartition du mercure ainsi corrigé confirme la rivière Détroit comme source prédominante de mercure industriel dans le bassin ouest, avec transport subséquent à travers le bassin central et retombée finale dans le bassin est. Il y a évidence d'apports additionnels d'Erié, Pennsylvanie, et de Buffalo, New York. L'analyse du sédiment en suspension dans la rivière Détroit n'indique pas de changement de concentration entre 1970 et 1974, avec niveau moyen en 1974 de 1060 pp10^9. Nous estimons le chargement de mercure à 7.2 t.m./an dans les sédiments du bassin ouest. Nous avons observé une relation entre le mercure total et le cargone organique montrant deux régressions distinctes: 'A' avec une basse teneur en mercure à lien organique, représente les niveaux liés aux chargements industriels et naturels existant avant l'introduction d'usines cloralcali de cellules de mercure; 'B' applicable à des sédiments fortement contaminés, indique des changements dérivés de l'industrie des chloralcalis.

Accepted November 17, 1975

Accepté le 17 novembre 1975

A commercial fishing ban on walleye and white bass was imposed by the Ontario Provincial Government on fisheries in the Western basin of Lake Erie in June 1970, after the discovery that certain species contained mercury in excess of the 0.5 ppm acceptable level for human consumption (Leach and Nepszy 1976). This followed the March closure of commercial fishing in Lake St. Clair due to high levels of mercury in the St. Clair River system, attributed to industrial mercury losses from a Chlor-Alkali plant in Sarnia, Ont. During and subsequent to this period, many studies were carried out on fish mercury levels in the region by Provincial, State, and Federal agencies (e.g. F.W.Q.A. 1970). These studies were expanded to include the analysis of both water and sediment.

The significance of sediment-bound mercury relative to the aqueous ecosystem was elucidated in part in Minamata Bay, Japan, (Fujiki 1963)

Printed in Canada (J3994)
Imprimé au Canada (J3994)

following the well known and documented occurrence of mercury poisoning (Minamata Disease), in Sweden by Jensen and Jernelöv (1969), and in studies at the University of Illinois in the USA (Wood 1972). In essence it was demonstrated that inorganic mercury contained in bottom sediments could be transformed through bacterial activity to soluble methylmercury compounds which are taken up both by benthic and pelagic organisms. Biomagnification within food chains and cumulative uptake with growth were observed in fish in areas of mercury contamination (D'Itri 1971; Jernelöv 1970; and Håkanson 1975).

The surficial sediments of Lake Erie, as described here, were sampled in the summer of 1971 as a part of a systematic grid sampling program of the Great Lakes (Thomas et al. 1976). Samples recovered were analyzed for total mercury to establish the concentration levels and to determine source and dispersion pathways on a regional lake basis.

Detailed evaluation of mercury concentrations in a large number of sediment cores and surface sediments taken from the Western basin have been given by Walters et al. (1972), Kovacik and Walters (1973), Walters and Herdendorf (1973), Walters et al. (1974a), and Wolery and Walters (1974). These authors were able to show that the Detroit River constituted the primary source of mercury which was distributed within the basin relative to three identifiable water masses derived from the river. They ascribed pollutant mercury input to two chlor-alkali facilities located at Wyandotte, Mich., and Sarnia, Ont. (Wolery and Walters 1974, and Fig. 1). These latter authors further contended that increases in mercury concentrations in sediment cores could be related to the commencement of specific industrial activities and utilized this factor in computing rates of sedimentation for the Western Basin. Additional analyses of Lake Erie sediments on a more restricted basis have been given by the F.W.Q.A. (1970), Sivasankara et al. (1971), and Skoch and Turk (1972).

Levels of mercury in Lake Erie waters have ben discussed by Chau and Saitoh (1973). These authors indicate a total mercury average slightly above 0.13 μg/liter. They also observed that high mercury concentrations in water occurred in low turbidity water and suggested that, under higher turbidity levels, mercury was scavenged by the suspended particulates. They concluded that 30–90% of the total mercury in the Great Lakes waters is tightly held by suspended matter, thereby indicating the importance of suspended particulates in the transportation of mercury in these lake systems.

Methods

Sampling methods together with the sampling grid pattern (Fig. 1) are described by Thomas et al. (1976).

Subsamples for geochemical analysis were freeze-dried after recovery, ground to 100 mesh, and homogenized. An aliquot of each of the sample powders was submitted to Barringer Research Ltd.,[1] for total Hg analysis, using the combustion procedure developed by Barringer (1966).

It has been suggested that some loss of mercury by volatilization is induced by the freeze-drying method of sediment sample preparation. The loss may be in the order of 20% (Kennedy et al. 1971). However, no loss on freeze-drying was observed by Sivasankara et al. (1971). Freeze-dried samples were used in this study to maintain consistency with data from the remainder of the Great Lakes (Thomas 1972, 1973). However, it should be noted that the Hg values quoted in this paper may well be low and could represent only 80% of the probable true values.

Other physical and chemical parameters used in this study were analyzed by the following methods: total iron and aluminum calculated as Fe_2O_3 and Al_2O_3 by X-ray fluorescence spectrometry (Philips' automatic 1220C spectrometer); organic carbon and inorganic carbon by an induction furnace (Leco) carbon analyzer and by sulfurous acid digestion at room temperature; total quartz by potassium pyrosulphate fusion after the method of Trostell and Wynne (1940). The percentage of gravel, sand, silt, and clay size materials, based on the Wentworth classification, were derived from long pipette and sieve analyses (Thomas et al. 1972, 1973).

Results and Discussion

Total Mercury Abundance and Distribution

Thomas et al. (1976) have discussed in some detail the distribution of sediment types in Lake Erie, together with textural parameters and their association with the defined basins of sedimentation.

In general, the distribution patterns of total mercury in the sediments show conformity to the sediment distribution, with higher values occurring in the finer basin deposits and with lower values associated with the coarser sediment of the cross-lake sills and in the Inshore Zone (Fig. 2). The highest values occur in the Western Basin associated with the Detroit River.

The distribution patterns observed show agreement with those given by Walters et al. (1972, 1974a). The calculated mean value for the surface material of 63 cores analyzed by Walters et al. (1974a) was 880 ppb compared with 1217

[1]Barringer Research Ltd., Carlingview Drive, Rexdale, Ont.

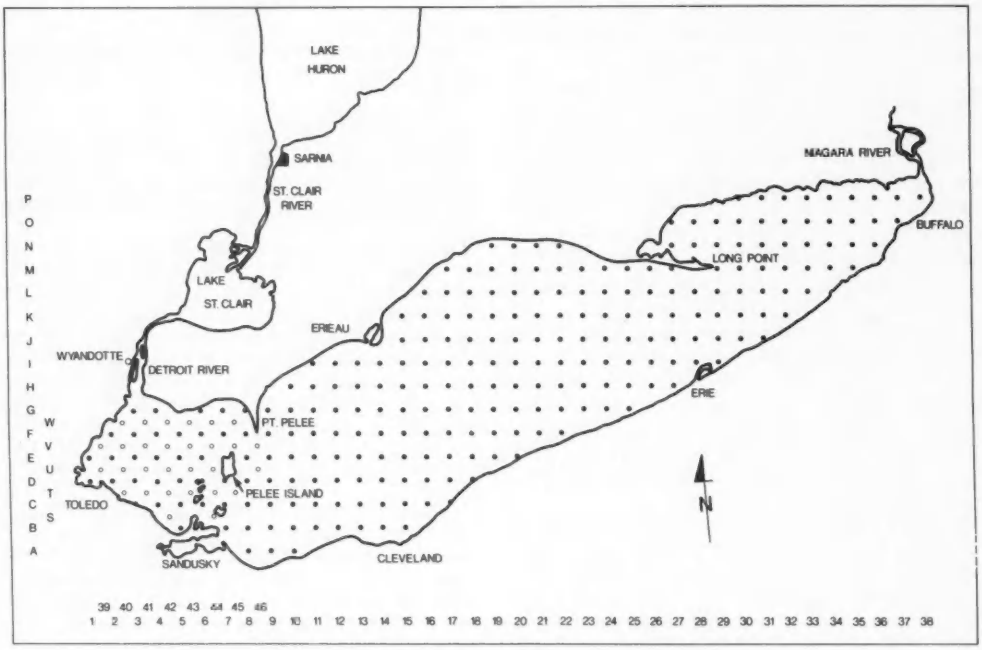

FIG. 1. Sample distribution plan including localities designated in text.

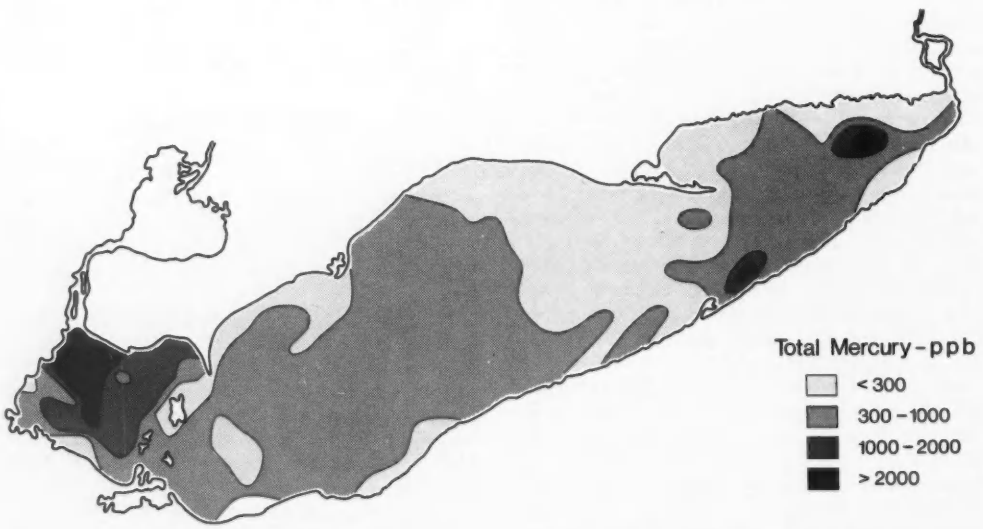

FIG. 2. Distribution of total mercury in the surficial 3 cm of sediment in Lake Erie.

ppb from 53 samples in the Western basin and associated Inshore Zone from this study (Table 1). A Student's *t*-test showed a significant difference between these means at the 95% confidence level yet, at the 99% level, the means were not significantly different.

Total mercury in Lake Erie sediments ranges from 8 to 2900 ppb ($\bar{x}$ 580 ± 560 ppb SD) (Table

TABLE 1. Mercury levels (means ± SD) in the sediments of Lake Erie (range in parentheses).

Basin	Mercury concentration (*ppb*)
Inshore Zone (whole lake)	287±323; N = 102 (8–1881)
Western Basin	1622±694; N = 34 (484–2929)
Western Basin with its inshore zone	1217±792; N = 53 (65–2929)
Sandusky Basin	710±464; N = 8 (271–1810)
Central Basin	544±191; N = 85 (56–1030)
Eastern Basin	483±272; N = 31 (45–977)
Total	582±555; N = 259 (8–2929)

TABLE 2. Surface sediment mercury concentrations (mean ± SD) in the Great Lakes (ranges in parentheses). Depth of sample 0–3 cm.

Lake		Mercury concentration (*ppb*)
Superior		83±56; N = 405 (4–584)
S. Michigan[1]		146±—; N = 31 (30–380)
Huron	North Channel	151±231; N = 55 (8–1112)
	Georgian Bay	257±1013; N = 117 (12–9500)
	Huron	222±162; N = 163 (54–805)
St. Clair		632±628; N = 55 (70–2565)
Erie		582±555; N = 259 (8–2929)
Ontario		651±507; N = 248 (32–2100)
Average Gt. Lakes		347±—; N = 1317 (4–9500)

[1]Depth of sample variable (Kennedy et al. 1971). Hg values taken for the topmost subsamples of the sediment cores. This subsample varied from 0–1 to 0–5 cm, but was generally the topmost 2 or 3 cm. SD not calculated.

1). Mean mercury values for the individual basins and the Inshore Zone are given in Table 1. This table shows that high values occur in the sediments of the Western basin compared to the remainder of the lake, with the mean values for the Eastern basin and the Central basin showing similar lower concentrations. In order to place the mercury sediment levels in Lake Erie in perspective with respect to the Great Lakes System, the total lake mean value may be compared with those from the other Great Lakes and given in Table 2 (Thomas 1974). This table shows that the lower Great Lakes of Erie and Ontario together with Lake St. Clair have mean values in the order of three times the values for Lake Huron and Georgian Bay. This indicates that industrial contamination in terms of flow-through of water masses in the Great Lakes starts between Lake Huron and Lake St. Clair. This observation, together with the distribution pattern of total mercury in Lake Erie sediments, points to the St. Clair/Detroit river system as being the predominant source of mercury, in agreement with the observations of Walters and his co-workers (see introduction).

QUARTZ-CORRECTED MERCURY DISTRIBUTION

Thomas (1972, 1973), in studies of the distribution of mercury in the sediments of Lakes Ontario and Huron, applied a quartz correction to the total mercury concentrations to evaluate the source and transport of sediment-bound mercury. This correction was based on the interpretation of statistical interrelationships observed in those lakes, which suggested that the mercury was partitioned between a number of mineral phases occurring in the finer fractions of the sediment. These included amorphous, hydrated Fe oxide, organic matter, and sulphide. Similarly, it can be assumed that quartz and feldspar, the major components of the coarser materials in Lake Erie sediments (Thomas et al. 1976), are inert with respect to mercury. The relationship of quartz to mean grain size has been noted by Thomas et al. (1976) implying dependency of texture on sediment quartz content.

Figure 3 shows that the relationship of total mercury to quartz is an inverse one, with the same patterns emerging as for the relationship of total mercury to Org. C as will be discussed later. The approximated regression line "A" expresses the trend for modern sediments moderately contaminated by Hg. The highest values on this line [zone (1)] are close to 1000 ppb and are found in the western part of the Central basin. Line "B" shows the dependence of Hg on quartz in highly contaminated samples, most of which are found in the Western basin. In spite of their high quartz content, samples falling in region (2) of the diagram are relatively highly contaminated, indicating a very strong adsorption of Hg in their fine fraction. A correction can, thus, be applied for these diluents, which gives an insight into the regional distribution and trends of mercury associated with the fines in the lake. A calculation may

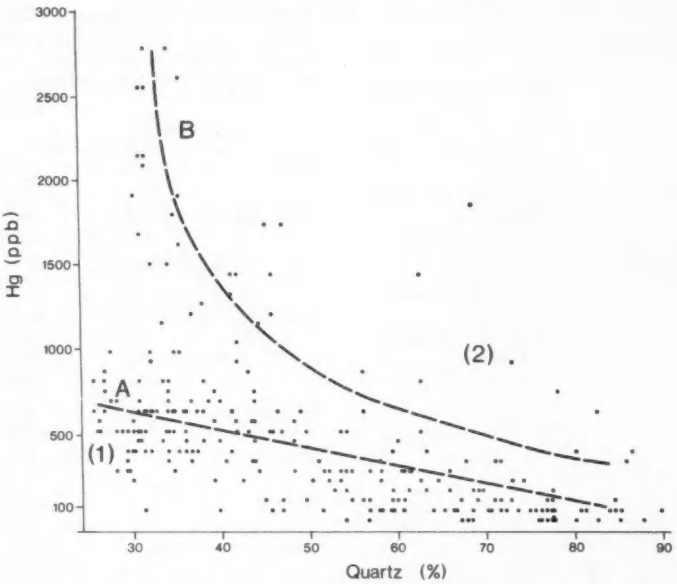

FIG. 3. Relationship of quartz to total mercury (*see text for explanation*).

thus be applied as follows:

$$\text{Hg (quartz corrected)} = \frac{\text{Hg (observed)} \times 100}{(100 - \text{quartz \%})}$$

It has been demonstrated (Thomas 1969, Thomas et al. 1976) that quartz is a very minor constituent in the less than 4 μ (8 Φ) grain-size fraction and that there is a direct correlation between mean grain-size and quartz content. As such, quartz-corrected mercury gives an assessment of the degree of mercury concentration or adsorption by the fine particulate materials in the lake sediments.

The distribution of quartz-corrected mercury is shown in Fig. 4, which indicates significant differences from the total mercury distribution given in Fig. 2. The quartz-corrected mercury concentration gradient in the Western basin again verifies the Detroit River as the primary source. Concentrations are more uniform in the Central basin but a clear anomaly of higher mercury can be seen extending from just east of Pelee Passage (south of Pelee Point, Fig. 1) eastwards to the south shore of the lake to the east of Erie, Pennsylvania. Higher values occur immediately to the north of Erie with a further anomaly west of Buffalo in the Eastern basin. In general terms this may be interpreted as indicating the major pathways for the transportation of sediment-bound mercury on a regional lake basis. From the Detroit River, sediment-bound mercury is transported fanwise into the Western basin. Concentration contours suggest that these materials are in part moving through Pelee Passage, presumably both north and south of Pelee Island into the Central basin. Here the distribution indicates movement eastwards to the south shore, with a continuation of transportation eastwards along the south shore to final deposition in the Eastern basin. The anomaly to the north of Erie may reflect additional mercury loading from Erie but may be more likely due to a concentration of mercury enriched fines reflecting water current reversals in this region [Simons 1976 (Fig. 3 and 4)].

A break in the anomaly extension from the Western to Central basins can be observed immediately to the east of Pelee Island, due to the occurrence of a zone of low quartz-corrected mercury concentrations aligned north to south from Pelee Point, almost to the southern shore of the lake. This low mercury zone overlays the Pelee moraine in a nondepositional area of lag gravels and sands. This low anomaly is indicative of the low mercury concentrations in fine particles of direct glacial origin. The anomaly in the Eastern basin is believed to be due to materials derived from Buffalo harbor where high values have been observed in sediment and plankton

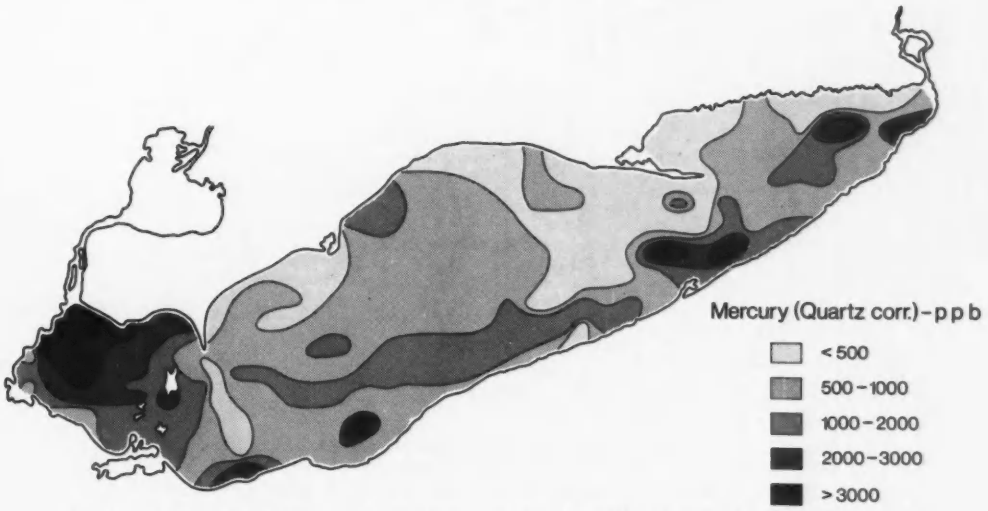

FIG. 4. Distribution of quartz-corrected mercury in the surficial 3 cm of sediment in Lake Erie.

(Sivasankara et al. 1971; and Walters et al. 1974b). In general summation, the distribution of quartz-corrected mercury reflects to a considerable degree the computed water circulation patterns of the lake (Simons 1976).

MERCURY SOURCES AND LOADINGS TO THE WESTERN BASIN

It has been previously noted that the major source of mercury to Lake Erie occurs as direct loading to the Western basin from the Detroit River, confirming the conclusions of Walters and Herdendorf (1973). During 1971, bottom sediment samples were taken from the mouth of the Detroit River and analyzed for mercury, quartz, and organic carbon as described previously. Figure 5A shows the total observed mercury levels at each sample location with values ranging from 0.05 to 5.0 ppm. The low values occur in areas of high flow whereas the higher values occur in regions of interrupted flow and in depositional zones occurring at the mouth of the river (Samples 0–2, 0–3, Fig. 5A). Quartz-corrected values (Fig. 5B) show that all samples for which values are available other than M-2 and N-2 have high values indicative of contamination of bottom sediment fines by upstream sources of mercury. The distribution of these values further suggests a relatively uniform distribution across the river with no evidence for a specific source on either bank of the river in the immediate upstream area. This implies that sources in addition to Wyandotte (west bank) are loading mercury into the Detroit River system. Analyses conducted by F.W.Q.A. (1970) upstream of Wyandotte on suspended particulates gave a value of 1.0 ppm. Further suspended solids analyses were conducted by R. L. Thomas (unpublished data) in 1974 on five samples between Lake St. Clair and the mouth of the Detroit River. Mercury values in these samples ranged from 0.97 to 1.11 ppm with an average of 1.06 ppm. Thus it is obvious that no apparent change has occurred in the mercury content of suspended sediment transported from Lake St. Clair to Lake Erie between 1970 and 1974. These limited observations suggest that Lake St. Clair is serving as a mercury-enriched sediment reservoir whereby resuspended sediment is being transported downstream to Lake Erie. This process is currently under further investigation.

The calculation of total annual loadings to the Western basin of Lake Erie, based on the data used in this study (53 surface samples), gives a value of 7.2 m.tons.yr^{-1} which compares very closely to the 6.9 m.tons.yr^{-1} given by Kemp et al. (1976). A calculation of annual loading from the data given by Walters and Herdendorf (1973) gives a range of 5.7–6.7 m.tons.yr^{-1}. This is in good agreement with the values given above, particularly as the lower value derived from the data of Walters and Herdendorf (1973) is based on an average for the past 35 yr of sediment accumulation. The annual loading predicted by Walters et al. (1974a) by averaging the total Hg reservoir over 31 yr is 7.4 m.tons.yr^{-1}.

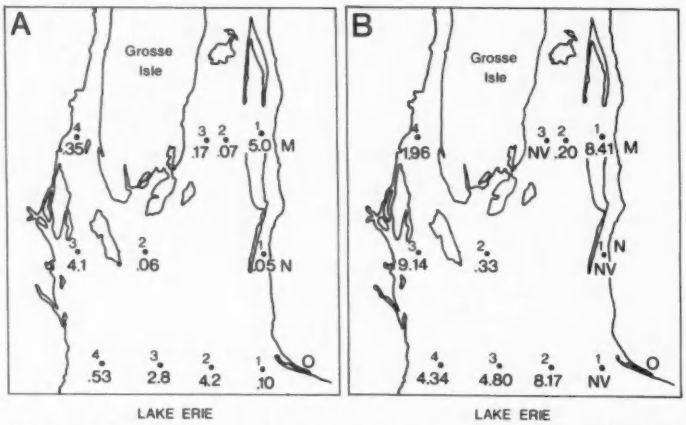

FIG. 5. Detroit River mouth samples; A, Sample nomenclature, lines lettered M, N, O from North to South with sequential numbering on each line from East to West. Values given are for total mercury. B, Quartz-corrected mercury (NV = no values).

INTERRELATIONSHIPS BETWEEN VARIABLES

The interrelationships observed between selected variables are summarized in the correlation coefficient matrix given in Table 3. A good relationship can be observed between observed total Hg and Org. C and a significantly lesser one between Hg and the various forms of Fe. On the basis of the data of Thomas (1972, 1973) this is interpreted as suggesting adsorption/complexation of mercury with organic matter and hydrated iron oxides. The association of mercury with organic matter has also been described by Cranston and Buckley (1972), Kennedy et al. (1971), and Walters et al. (1974b), and the association of Hg with hydrated iron oxides by Vernet and Thomas (1972). From Table 3 it can also be seen that both Org. C and iron show a high correlation with the percentage clay, confirming the association of Hg with the fine sediment material.

The relationship of mercury to Org. C is illustrated in the scatter diagram Fig. 6. As in Fig. 3, two factors emerge. First, trend line "A" shows a high degree of relationship of mercury to Org. C in samples with mercury concentrations below 906 ppb. Samples with mercury levels above 906 ppb show a wider scatter with a lesser relationship to Org. C concentration. Where a high degree of correlation exists, it can be stated that organic matter to environmental mercury concentrations are in equilibrium and that such a relationship represents the natural situation for the time interval represented by the samples. This is not to be inferred as a relationship representing the natural background levels, but rather the situation prior to the increase of mercury from the chlor-alkali industries of the St. Clair/Detroit river system. Second, trend line "B" is indicative of increased adsorption/complexation of mercury by organic matter related to specific events and localities. Designation of the samples in Zone B shows that they occur predominantly in the Western Basin and thus are likely to represent more recent influxes of industrial mercury contaminants.

TABLE 3. Correlation coefficients between selected variables (228 samples). All are signifficant at a 0.01 probability level.

	Hg	Clay	Org. C	Fe_2O_3	Al_2O_3
Hg	1.0				
Clay	0.41	1.0			
Org. C	0.60	0.81	1.0		
Fe_2O_3	0.29	0.49	0.52	1.0	
Al_2O_3	0.38	0.90	0.84	0.48	1.0

Summary and Conclusions

Mercury in the sediments of Lake Erie ranges from 8 to 2929 ppb with a mean of 582 ppb. The mean value is similar to the mean values observed in Lakes St. Clair and Ontario and approximately three times the mean value for Lake Huron. Highest values are observed in the Western basin with lower similar values occurring in the Central and Eastern basins.

The distribution of total mercury indicates 1) high levels of mercury in the Western Basin asso-

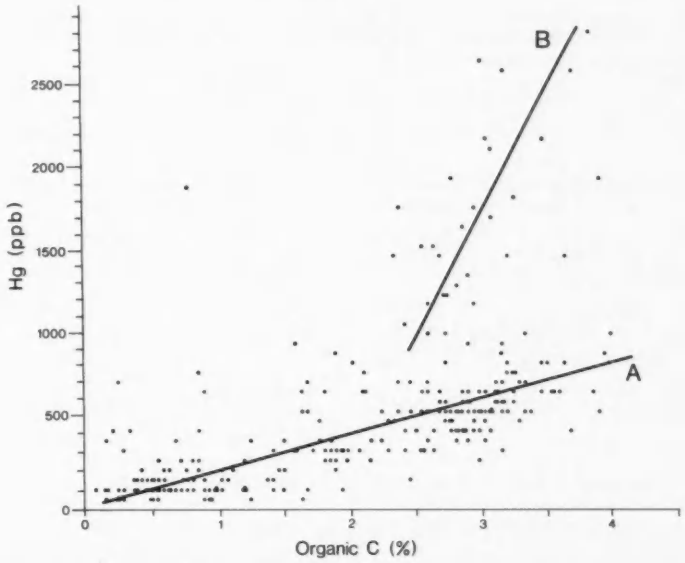

FIG. 6. Relationship of total mercury to Org. C (*see text for explanation*).

ciated with the Detroit River and 2) an overall distribution related to the texture of the lake sediments, with higher levels in the basin muds relative to the coarser-textured inshore zone and cross-lake moraines.

Quartz-corrected mercury distributions were used in an attempt to define source and dispersal of mercury-contaminated sediment. The distribution shows conclusively that the major source of mercury is the Detroit River. It also indicates that the dispersal of mercury-contaminated sediments occurs fanwise from the Detroit River mouth with some transfer via Pelee Passage to the Central basin. In the Central basin, the distribution of quartz-corrected mercury showed good agreement with computed water circulation suggesting final deposition in the deep water of the Eastern basin. Quartz-corrected mercury also suggested some input loadings of mercury to the Eastern basin from Erie, Pa., and Buffalo, N.Y.

Statistical evaluation demonstrated that mercury in Lake Erie has a direct relationship to Org. C. Two relationships with Org. C, designated "A" and "B" were observed. Regression "A" is interpreted as representing an "Equilibrium Condition" reflecting levels of organic-bound mercury stabilized to industrial and natural loadings prior to more recent large loadings derived from mercury cell chlor-alkali plants. Regression "B" reflects organic-bound mercury levels, particularly in the Western basin, associated with more recent increased loadings noted above.

Acknowledgments

We thank our many colleagues at C.C.I.W. for helpful discussion during the period of this study. Thanks are expressed particularly to T. W. Morton for analytical support and to the crew of the C.S.S. *Limnos* for their enthusiasm and support during the field operations.

BARRINGER, A. R. 1966. Instrument for high sensitivity mercury analysis of soils, rocks and air. Trans. Inst. Min. Met. (London) 75: B120–B124.

CHAU, Y. K., AND H. SAITOH. 1973. Determination of methyl mercury in lake water. Int. J. Environ. Anal. Chem. 3: 133–139.

CRANSTON, R. E., AND D. E. BUCKLEY. 1972. Mercury pathways in a river and estuary. Environ. Sci. Technol. 613: 274–278.

D'ITRI, F. M. 1971. The environmental mercury problem. Mich. State Univ. Inst. Water Res. Tech. Rep. 12: 289 p.

FEDERAL WATER QUALITY ADMINISTRATION. 1970. Investigation of mercury in the St. Clair River–Lake Erie system. Open File Rep., 108 p.

FUJIKI, M. 1963. Studies on the course that the causative agent of Minamata disease was formed, especially on the accumulation of the mercury compound in the fish and shellfish of Minamata Bay. J. Kumamoto Med. Soc. 39–494.

HÅKANSON, L. 1975. Kvicksilver i Vänern — nulage och prognos (Mercury in Lake Vänern — present status and prognosis) ų Statens Naturvårdsverk PM 563, NLU Rapport 80: 121 p.

JENSEN, S., AND A. JERNELÖV. 1969. Biological methylation of mercury in aquatic organisms. Nature 223: 753–754.

JERNELÖV, A. 1970. Release of methyl mercury from sediments with layers containing inorganic mercury at different depths. Limnol. Oceanogr. 15: 958–960.

KEMP, A. L. W., R. L. THOMAS, C. I. DELL, AND J.-M. JAQUET. 1976. Cultural impact on the geochemistry of sediments in Lake Erie. J. Fish. Res. Board Can. 33: 440–462.

KENNEDY, E. J., R. R. RUCH, AND N. F. SHIMP. 1971. Distribution of mercury in unconsolidated sediments from southern Lake Michigan. Environ. Geol. Notes Ill. State Geol. Surv. 44.

KOVACIK, T. L., AND L. J. WALTERS. 1973. Mercury distribution in sediment cores from western Lake Erie. Proc. Conf. Great Lakes Res. 16: 252–259.

LEACH, J. H., AND S. J. NEPSZY. 1976. The fish community in Lake Erie. J. Fish. Res. Board Can. 33: 622–638.

SIMONS, T. J. 1976. Continuous dynamical computation of water transports in Lake Erie for 1970. J. Fish. Res. Board Can. 33: 371–384.

SIVASANKARA, P. K., C. C. THOMAS, J. A. SONDEL, AND C. M. HYCHE. 1971. Determination of mercury in biological and environmental samples by neutron activation analysis. Anal. Chem. 43: 1419–1425.

SKOCH, E. J., AND J. M. TURK. 1972. Fluctuations in the level of mercury in sediments collected from the island area of Lake Erie, 1964 to 1968. Proc. Conf. Great Lakes Res. 15: 291–297.

THOMAS, R. L. 1969. The qualitative distribution of feldspars in surficial bottom sediments from Lake Ontario. Proc. Conf. Great Lakes Res. 12: 364–379.

1972. The distribution of mercury in the sediments of Lake Ontario. Can. J. Earth Sci. 9: 636–651.

1973. The distribution of mercury in the surficial sediments of Lake Huron. Can. J. Earth Sci. 10: 194–204.

1974. The distribution and transport of mercury in the sediments of the Laurentian Great Lakes system. Proc. Int. Conf. Transport of Persist. Chemicals in Aquatic Ecosystems, Ottawa, 1.1–1.16.

THOMAS, R. L., A. L. W. KEMP, AND C. F. M. LEWIS. 1972. Distribution, composition and characteristics of the surficial sediments of Lake Ontario. J. Sediment. Petrol. 42: 66–84.

1973. The surficial sediments of Lake Huron. C. J. Earth Sci. 10: 226–271.

THOMAS, R. L., J.-M. JAQUET, A. L. W. KEMP, AND C. F. M. LEWIS. 1976. Surficial sediments of Lake Erie. J. Fish. Res. Board Can. 33: 385–402.

TROSTELL, L. J., AND D. J. WYNNE. 1940. Determination of quartz (free silica) in refractory clays. J. Am. Ceram. Soc. 23: 18–22.

VERNET, J.-P., AND R. L. THOMAS. 1972. Levels of mercury in the sediments of some Swiss lakes, including Lake Geneva and the Rhone river. Eclogae Geol. Helv. 65: 293–306.

WALTERS, L. J., C. E. HERDENDORF, L. J. CHARLESWORTH, H. K. ANDERS, W. B. JACKSON, E. J. SKOCH, D. K. WEBB, T. L. KOVACIK, AND C. S. SIKES. 1972. Mercury contamination and its relation to other physico-chemical parameters in the western basin of Lake Erie. Proc. Conf. Great Lakes Res. 15: 306–316.

WALTERS, L. J., AND C. E. HERDENDORF. 1973. Mercury concentration in surface sediments as related to water masses in western Lake Erie. Compass 50: 5–10.

WALTERS, L. J., T. L. KOVACIK, AND C. E. HERDENDORF. 1974a. Mercury occurrence in sediment cores from western Lake Erie. Ohio J. Sci. 74: 1–19.

WALTERS, L. J., T. J. WOLERY, AND R. D. MYSER. 1974b. Occurrence of As, Cd, Co, Cr, Cu, Fe, Hg, Ni, Sb and Zn in Lake Erie sediments. Proc. Conf. Great Lakes Res. 17: 219–234.

WOLERY, T. J., AND L. J. WALTERS. 1974. Pollutant mercury and sedimentation in the western basin of Lake Erie. Proc. Conf. Great Lakes Res. 17: 235–249.

WOOD, J. M. 1972. Environmental pollution by mercury, p. 39–56. *In* R. L. Metcalf and J. N. Pitts [ed.] Advances in environmental science and technology, Vol. 2. Wiley-Interscience, New York, N.Y.

Forms of Phosphorus in the Surficial Sediments of Lake Erie

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WILLIAMS, J. D. H., J-M. JAQUET, AND R. L. THOMAS. 1976. Forms of phosphorus in the surficial sediments of Lake Erie. J. Fish. Res. Board Can. 33: 413–429.

The phosphorus in 48 surficial Lake Erie sediment samples was present in three major forms: phosphorus associated with apatite, nonapatite inorganic phosphorus (NAIP), and organic phosphorus. The apatite was of natural, detrital origin. It existed as particles ranging from fine sand to clay in size but mostly as silt-sized particles and was concentrated in nearshore sediments. Both NAIP and organic phosphorus was concentrated in fine-grained sediments accumulating in offshore depositional areas. NAIP was associated with amorphous hydrated ferric oxide in the oxidized microzone but was present as vivianite ($Fe_3(PO_4)_2 \cdot 8H_2O$) and possibly other forms also in the reduced zone. The organic phosphorus content of the sediment was closely related to organic carbon content. The phyllosilicate, organic matter, and reactive iron and manganese components of the sediments existed in intimate association.

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Le phosphore, dans 48 échantillons de sédiment superficiel du lac Erié, est présent sous trois formes principales: phosphore associé à l'apatite, phosphore inorganique non-apatite (PINA) et phosphore organique. L'apatite est d'origine naturelle, détritique. Elle se présente sous forme de particules dont la grosseur varie du sable fin à l'argile, mais surtout de la grosseur de particules limoneuses, et est concentrée dans les sédiments littoraux. Le PINA et le phosphore organique sont tous deux concentrés dans les aires de sédimentation du large. Le PINA est associé à l'oxyde ferrique hydraté dans la micro-zone oxydée, mais est présent sous forme de vivianite ($Fe_3(PO_4)_2 \cdot 8\,H_2O$) et possiblement aussi sous d'autres formes dans la zone réduite. La teneur en phosphore organique du sédiment est étroitement liée à celle du carbone organique. Le phyllosilicate, matière organique, et les composantes fer réactif et manganèse des sédiments se rencontrent en étroite association.

Accepted November 17, 1975 Accepté le 17 novembre 1975

OVER 80% of the phosphorus entering Lake Erie is sedimented within the lake basin (International Joint Commission 1969) and it is safe to assume that the proportion sedimented was as high or even higher in earlier years. Consequently, the history of phosphorus input into the lake can be reconstructed from the record of the sediment column, provided that certain factors tending to blur or distort this record are unimportant or can be allowed for. Changes in the rates of sedimentation in Lake Erie, over the last 150 yr or more, of natural and anthropogenic phosphorus have been discussed by Kemp et al. (1976). To date, however, there has been little information on the chemical forms that contain phosphorus in sediments. Information on the forms of phosphorus in terrestrial soils has proved useful in understanding how a soil releases phosphorus into, or takes up phosphorus from soil solution. Similarly, the results of this study have been used by Lam and Jaquet (1976) in the modelling of regeneration of phosphorus from Lake Erie sediments and its effect on the phosphorus values of overlying water.

A review of the likely modes of combination of phosphorus in lake sediments, and of analytical procedures for their determination, has been presented by Syers et al. (1973). In this and a companion paper (Williams et al. 1976), it is assumed that phosphorus in Lake Erie sediments can be subdivided into three major categories. The first category, designated "apatite phosphorus," consists of orthophosphate present in the crystal lattices of apatite grains. The second category, designated "nonapatite inorganic phosphorus" and abbreviated to NAIP, consists of all remaining orthophosphate ions. It is assumed that nonorthophosphate forms of inorganic phosphorus, in particular pyrophosphate and other polyphosphate ions, are absent from sediments or else present in only minor amounts. The NAIP category also includes soluble orthophosphate ions in the sediment pore waters. The last category is designated

Printed in Canada (J3995)
Imprimé au Canada (J3995)

"organic phosphorus" and embraces all phosphorus associated with carbon atoms via C-O-P or C-P bonds. It includes organic phosphorus compounds in the pore waters. This three-part classification resembles that proposed for phosphorus in terrestrial soils by Williams and Walker (1969), except that no distinction is attempted between different forms of NAIP here.

In addition to classification by chemical characteristics, sedimented phosphorus can also be categorized as being of detrital or nondetrital origin. "Detrital" is used here in its geological sense, i.e. detrital phosphorus in the sediments is phosphorus which is in essentially the same chemical and mineralogical forms as in the rocks, soils and unconsolidated surface deposits from which the sediments were derived. Most of the detrital materials entering Lake Erie are derived from eroding bluffs along the north shore of the Central basin (Kemp et al. 1976). Nondetrital phosphorus in the sediments includes all forms in which the phosphorus was at some earlier stage in solution in the lake or inflowing streams and rivers, the phosphorus being subsequently converted into particulate form by, for example, uptake by organisms or reaction with metal oxides. The nondetrital category also includes "diagenetic phosphorus," i.e. phosphorus present in new modes of combination which are created within the sediment column itself. An example is vivianite ($Fe_3(PO_4)_2 \cdot 8H_2O$), which is readily oxidised by air (Nriagu 1972) and hence can be formed only in the reduced zone of the sediment column.

Materials and Methods

During July 1971, samples of Lake Erie sediments were recoevred at 260 locations (Thomas et al. 1976). The materials collected represent the top 3 cm of the sediment column. From this set of samples a subset of 48 samples was chosen for this study, so as to cover all parts of the lake (Fig. 1). The proportion of samples with mean grain-size values falling within any pair of limits was approximately the same in both set and subset. Finally, two samples (W40 and M27) with unusually high total phosphorus content were also included.

The procedures used to determine the three categories of sediment phosphorus recognized here (apatite phosphorus, NAIP, and organic phosphorus) are outlined diagrammatically in Fig. 2. The estimates of each category that are obtained by these analytical procedures are denoted here by the expressions apatite-P, NAI-P, and organic P, in an attempt to maintain a rigid terminological distinction between the concept or physical reality and the analytical estimate. The method used to determine organic P, apart from minor modifications to fit the normal working day, was essentially that of Sommers et al. (1972). This procedure resembles others used for many years to determine the total organic phosphorus content of terrestrial soils. The precision and accuracy of these procedures have been discussed by Anderson (1960) and Williams et al. (1970). It is assumed that labile organic phosphorus compounds capable of undergoing hydrolysis during the analytical procedure, such as glucose-1-phosphate (Robertson 1958), are also too reactive to accumulate in soils or lake sediments in appreciable amounts.

The sequential extraction procedure used to estimate NAI-P and apatite-P (Fig. 2) is derived from similar approaches used in attempts to identify the forms of inorganic phosphorus in terrestrial soils and lake sediments (Williams and Mayer 1972). The rationale behind the procedure will be discussed elsewhere (J. D. H. Williams, in preparation). Part of the rationale is based on the findings of this study.

The sum of the apatite-P, NAI-P, and organic P values was essentially equal to the total amount of phosphorus extracted from the sediment sample used to determine organic P. The expression "total P" was used to denote the total extracted. This value was slightly less than the true value of the total phosphorus content of the sediments, the difference being due to forms of phosphorus not extracted during the organic P procedure. The expression "inorganic P" was used to denote the amount of orthophosphate extracted during the same procedure. Inorganic P was essentially equal to apatite-P plus NAI-P.

The total iron content of the sediments was determined by X-ray fluorescence spectrometry. These values are denoted by the expression "Fe_T." A measure of the amount of iron in the sediment that was extractable by vigorous acid treatment was obtained by digesting 2.5 g of sample with 40 ml of concentrated hydrochloric acid for 30 min at 90 C. Iron in the digest was determined by atomic absorption spectrophotometry. The resulting values are denoted by the expression "conc. HCl-Fe." The amounts of total and acid-extractable manganese ("Mn_T" and "conc. HCl–Mn") were determined similarly. The organic carbon contents of the sediments ("organic C") were determined by the procedure of Kemp and Lewis (1968). The percent content of sand, silt, and clay in the samples, and the mean grain size expressed in ϕ units (where $\phi = -\log_2$ [mean particle size in mm]), were determined by standard sedimentological methods (Thomas et al. 1972). The last parameter is abbreviated to "mean ϕ" here. The separate analytical values for each sample are presented in Table 1, and the means, standard deviations and coefficients of variation of each parameter are summarized in Table 2. Values of conc. HCl–Fe and conc. HCl–Mn for sample H27 were not available.

The forms of the relationships between any parameter compared in turn with all other parameters were initially inspected by computer-drawn scatter diagrams. These revealed that, in the case of 43 out of 48 samples, all parameters were related to each other in a "regular" manner, i.e. the points lay within a fairly well defined region, though the degree of scatter varied greatly depending on which two parameters were being compared (see Fig. 4, 5, 7, and 8).

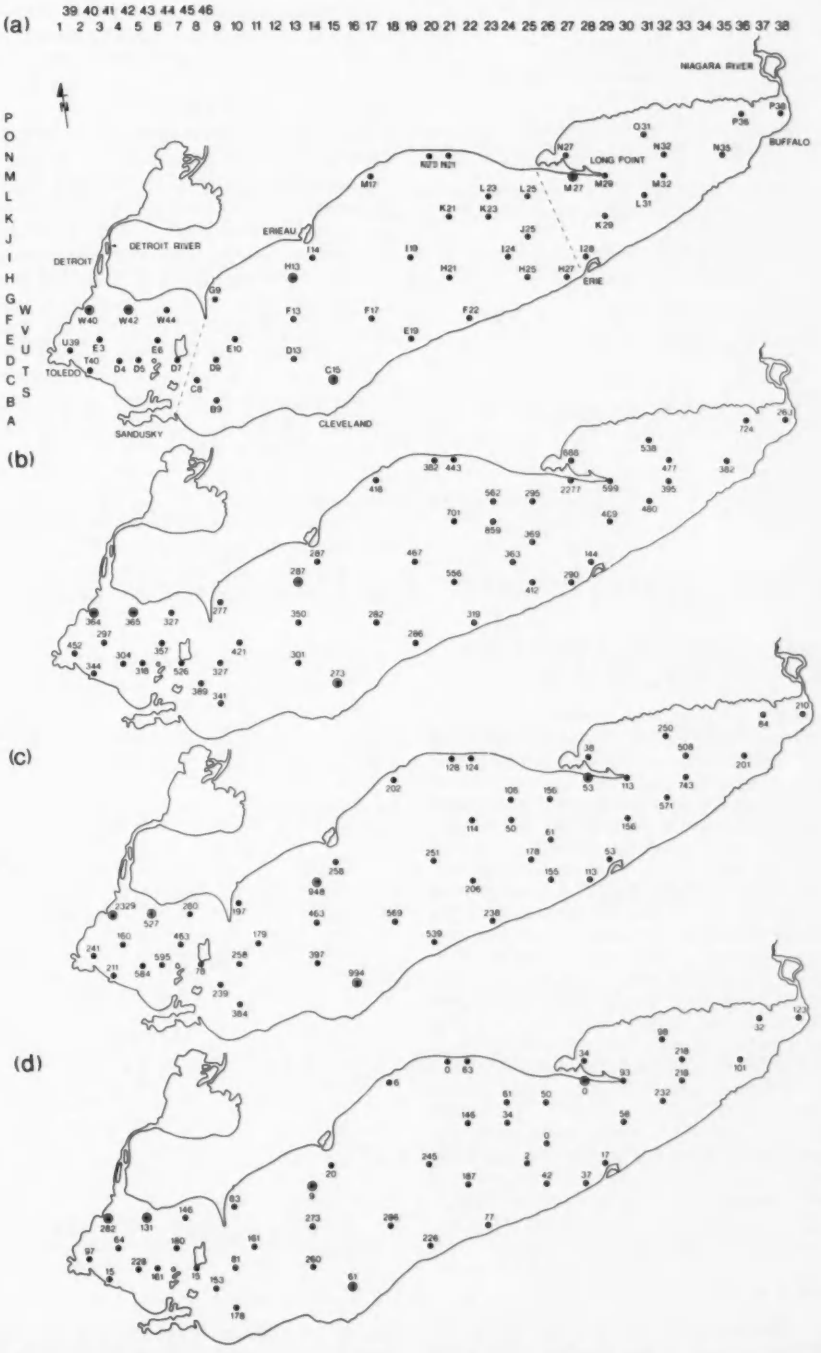

FIG. 1. (a) Location of sample sites. Values at each site of (b) apatite-P, (c) NAI-P, and (d) organic P. Dashed lines indicate boundaries between the Western, Central, and Eastern basins.

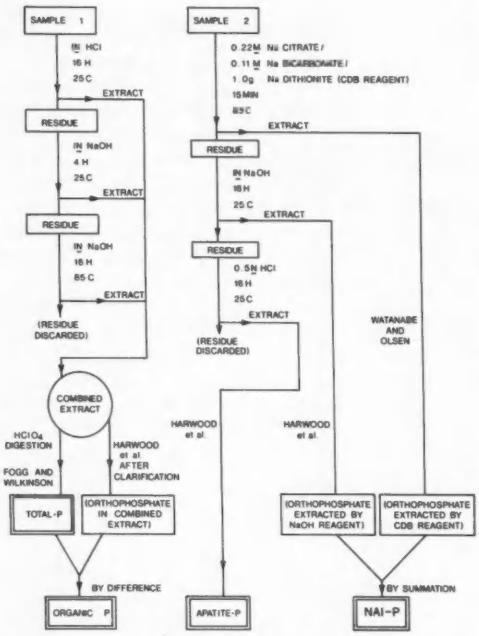

FIG. 2. Outline of analytical procedures used to determine apatite-P, NAI-P, and organic P.

Where this was the case, the relation between the two parameters was termed "normal" for these samples. In most instances, a small number of points, not more than four for any given comparison, lay well outside the reigon in which the remaining points were concentrated and did not conform to the trend followed by the majority of the points. In this case the relationship between the two parameters was termed "abnormal" for the sample or samples concerned. Only five out of the 48 samples displayed "abnormal" relationships no matter what parameters were being compared: these were samples W40, W42, H13, C15, and M27 (Fig. 1). Reasons for the unusual behavior of these samples will be discussed as the question arises. Table 3 lists the correlation coefficients relating to each pair of parameters in turn for the 43 samples which behaved in a "normal" manner in all comparisons. Coefficients whose moduli were greater than 0.29, 0.38, and 0.47 were significant at the 5, 1, and 0.1% levels, respectively. For comparison, the correlation coefficients, generally much lower, obtained when all available pairs of values were compared are also included in Table 3.

Results

Areal Distribution and Sources

Owing to the variability in the values of apatite P, NAI-P and organic P the number of samples analysed was not sufficient to demonstrate lake-wide trends diagrammatically; however, apatite-P tended to be higher in nearshore regions, especially in a region extending WSW from Long Point, while NAI-P and organic P values tended to be higher in offshore regions (Fig. 1).

The histogram of apatite-P values, expressed as $mg \cdot kg^{-1}$ dry sediment (or ppm) (Fig. 3) indicates that over half the samples had values in the range 250–400 $mg \cdot kg^{-1}$, but the distribution of values was somewhat skewed. Three of the four samples containing more than 700 $mg \cdot kg^{-1}$ were found in the northeastern sector of the Central basin, and one (M27) contained over 2000 $mg \cdot kg^{-1}$ apatite-P. These findings suggest that detrital apatite, derived from eroding bluffs along the northern shore of the Central basin (Williams et al. 1976), is concentrated in this region. The average value of apatite-P in the 48 samples (445 $mg \cdot kg^{-1}$) (Table 2) is within the range (410–610 $mg \cdot kg^{-1}$) for 14 bluff samples from this area (Williams et al. 1976). Apatite phosphorus also accounts for a very variable proportion of the total phosphorus (Fig. 3).

Kemp et al. (1976) have shown that most of the material sedimented in the lake is derived

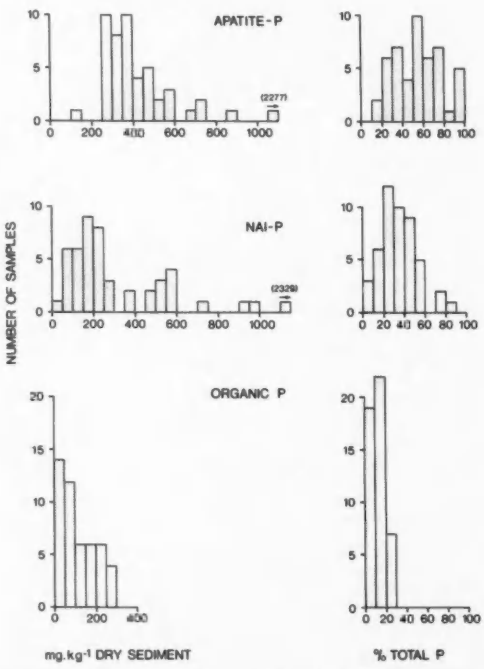

FIG. 3. Histograms of apatite-P, NAI-P, and organic P, expressed as $mg \cdot kg^{-1}$ dry sediment or as a percentage of total P.

TABLE 1. Individual values of each parameter for each sample.

Sample no.	Apatite P	NAI-P	Organic P	Inorganic P	Total P	Fe_T	Conc. HCl–Fe	Mn_T	Conc. HCl–Mn	Organic C	Mean	sand	silt	clay
	($mg \cdot kg^{-1}$)					(%)		($mg \cdot kg^{-1}$)		(%)	ϕ	(%)		
U39	452	241	97	689	786	2.40	1.73	540	389	1.76	5.48	41.9	34.8	43.2
T40	344	190	15	546	561	2.56	1.94	540	343	0.86	4.96	40.1	26.3	32.4
W40	364	2329	282	2581	2863	5.19	4.16	930	1022	4.44	7.51	1.2	50.1	48.6
E3	297	160	64	477	541	1.99	1.46	390	349	0.85	2.78	87.6	5.0	7.3
D4	304	584	228	916	1144	5.08	4.12	930	800	2.79	8.84	0.0	31.0	68.8
W42	365	506	131	897	1028	4.42	3.51	700	622	3.69	8.29	1.0	40.3	58.6
D5	318	595	161	935	1096	4.76	4.00	930	780	2.65	8.33	0.0	46.2	53.7
E6	357	463	180	773	953	4.11	3.11	770	626	2.91	7.59	5.1	42.1	52.7
W44	327	280	146	595	741	3.45	2.59	620	681	2.98	7.10	3.3	56.0	40.6
D7	526	78	15	564	579	3.22	2.26	620	349	0.76	1.01	94.8	3.2	1.9
C8	389	239	153	651	804	3.82	3.32	770	636	2.28	7.19	5.3	46.4	48.1
B9	341	384	178	727	905	4.35	3.32	700	523	2.78	8.06	0.7	39.3	59.9
D9	327	237	81	561	642	3.21	2.66	770	632	1.36	0.09	89.2	6.7	3.9
G9	277	197	83	481	564	3.21	2.49	770	623	1.40	−0.32	93.5	3.9	2.5
E10	421	179	161	625	786	3.08	2.24	620	482	2.81	6.43	25.3	33.4	41.2
D13	301	397	260	723	983	5.25	4.07	850	653	3.20	9.02	0.0	22.3	77.6
F13	350	463	273	814	1087	5.02	3.82	850	789	3.57	8.91	0.7	23.7	75.5
H13	287	948	9	1090	1099	7.69	6.57	2790	2881	1.04	0.86	88.9	5.9	5.0
I14	287	237	20	527	547	3.38	2.89	1160	982	0.32	1.68	97.6	1.4	0.9
C15	273	994	61	1325	1386	6.83	6.17	1080	934	3.65	6.10	26.5	45.4	27.9
F17	282	569	286	879	1165	5.29	4.10	1010	759	3.97	8.64	0.4	30.2	69.3
M17	418	181	6	577	583	3.71	2.99	930	746	0.51	−0.01	90.3	4.0	5.5
E19	286	539	226	814	1040	5.19	4.21	930	698	3.27	7.26	11.7	42.8	45.3
I19	467	251	245	678	923	4.33	3.24	700	555	2.89	7.82	0.9	44.6	54.3
N20	382	128	0	515	511	3.20	2.20	850	544	0.99	0.23	92.3	3.5	4.3
H21	556	206	187	731	918	3.36	2.53	620	517	2.44	7.13	7.6	51.3	41.0
K21	701	114	146	772	918	2.81	2.33	540	380	1.85	5.49	3.9	73.3	22.7
N21	443	124	63	538	601	2.56	1.80	770	718	1.24	4.72	51.4	17.4	31.1
F22	319	238	77	545	622	4.11	3.18	620	454	1.96	4.98	38.6	35.6	25.7
K23	859	50	34	905	939	2.36	1.88	620	350	0.89	4.57	41.9	48.2	9.7
L23	562	108	61	674	735	3.15	2.14	700	446	1.47	6.35	14.1	50.2	35.5
I24	363	178	2	500	502	3.16	2.16	1010	635	0.36	6.83	22.7	27.5	49.6
H25	412	155	42	580	622	3.13	2.21	540	412	1.50	5.33	43.3	24.6	32.0
J25	369	61	0	459	442	2.68	2.02	620	800	0.49	−2.09	100.0	0.0	0.0
L25	295	156	50	470	520	2.57	1.81	1080	772	1.17	−0.53	96.6	2.0	1.3
H27	290	113	37	402	439	2.93	n.d.	460	n.d.	0.70	3.07	91.4	5.3	3.1
M27	2277	53	0	2188	2176	9.59	2.67	2630	688	0.52	3.13	96.4	1.5	2.0
N27	688	38	34	719	753	1.83	1.26	620	544	1.44	5.55	6.2	82.6	11.0
I28	144	53	17	171	188	0.94	0.86	150	91	0.12	1.04	92.7	5.6	1.6
K29	489	156	58	659	717	3.11	2.63	770	580	0.94	6.73	23.7	27.7	48.4
M29	599	113	93	693	786	2.57	1.86	1010	962	1.65	6.91	2.1	66.7	31.1
L31	480	571	232	1101	1333	5.01	3.03	1780	1349	2.47	8.83	0.5	26.8	72.6
O31	538	250	98	852	950	4.31	3.27	1240	1305	1.64	8.35	1.4	39.6	58.9
M32	395	743	218	1169	1387	5.11	3.60	1010	822	3.02	8.56	0.0	34.1	65.8
N32	477	508	218	1037	1255	4.72	3.20	1630	1375	2.91	8.22	0.1	41.5	58.3
N35	382	201	101	557	658	3.66	2.55	540	390	1.90	6.72	0.8	66.6	32.5
P36	724	84	32	772	804	2.41	1.73	620	376	1.11	4.39	59.1	31.1	9.6
P38	263	210	123	508	631	3.55	2.75	620	560	1.86	3.18	53.7	31.5	14.7

TABLE 2. Mean values and measures of the degree of variability of each parameter from 48 samples of surficial sediments from Lake Erie.

Parameter		Mean	Range	SD	CV (%)
Apatite-P	($mg \cdot kg^{-1}$)	445	144–2277	304	68
NAI-P:					
extracted by CDB	(")	317	34–2283		
extracted by NaOH	(")	13	0–46		
sum of extracts	(")	330	38–2329	372	113
Organic P	(")	110	0–286	89	81
Inorganic P	(")	770	171–2581	405	53
Total P	(")	879	188–2863	443	50
Fe_T	(%)	3.84	0.94–9.59	1.53	40
Conc. HCl–Fe	(%)	2.86	0.86–6.57	1.12	39
Mn_T	($mg \cdot kg^{-1}$)	873	150–2790	481	55
Conc. HCl–Mn	($mg \cdot kg^{-1}$)	700	91–2881	419	60
Organic C	(%)	1.90	0.12–4.44	1.11	58
Mean grain size	(ϕ)	5.23	−2.09–+9.02	3.10	59
Sand	(%)	36.4	0.01–100	38.6	106
Silt	(%)	30.9	0–82.6	21.0	68
Clay	(%)	33.1	0–77.6	24.5	74

from eroding bluffs along the north shore of the Central basin. The tendency towards higher apatite-P values in the region south and west of Long Point suggests that apatite derived from these bluffs is preferentially deposited in this region; however, Fig. 1b also shows that apatite is present in all samples from this lake. It is unlikely that the dimensions of the finest-grained apatite particles are altered appreciably by water action within the basin, but clay-sized apatite may have been formed by ice abrasion of the primary source materials with rock flour production. Sample M27, off Long Point, had an anomalously high apatite-P value and high Fe_T and Mn_T values, but only about a quarter of the iron and the manganese was extractable by acid. This sample probably has a high heavy mineral content, and is the product of intense sorting due to the vigorous currents which sweep along the southern shore of Long Point (Lewis 1966).

In contrast to apatite-P, the NAI-P values were generally highest in the regions of deposition of fine-grained sediment, especially in the deepest part of the Eastern basin, east of Long Point (Thomas et al. 1976). The histograms of NAI-P values (Fig. 3) indicates a somewhat bimodal distribution, with the large group of samples averaging around 200 $mg \cdot kg^{-1}$ and a smaller group averaging around 500 $mg \cdot kg^{-1}$. Three of the five exceptional samples (W40, H13, and C15) contained over 900 $mg \cdot kg^{-1}$, the first sample over 2000 $mg \cdot kg^{-1}$. The NAI-P fraction accounted for 20–50% of the total phosphorus in most cases.

The organic P values showed a skewed distribution (Fig. 3), and averaged 112 $mg \cdot kg^{-1}$ for all samples (Table 2). No organic P value exceeded 300 $mg \cdot kg^{-1}$ or 27% of total P. The organic P values were generally lower in samples from nearshore areas and were zero in three cases. The fine-grained sediments from the Eastern basin had organic P values exceeding 200 $mg \cdot kg^{-1}$.

The conc. HCl–Fe values of the sediments averaged about 74% of Fe_T (Table 2). Much of the iron not extracted by acid was presumably present in unweathered minerals such as primary ferromagnesian silicate minerals, magnetite, ilmenite, etc. On average, 80% of the total manganese was extracted by concentrated hydrochloric acid (Table 2). It is thought that the non-extractable manganese was also mainly present in unweathered primary silicate minerals. With one sample (M27) excluded, conc. HCl–Fe and Fe_T were linearly related, with a correlation coefficient, r, equal to 0.974. Similarly, with the same sample excluded, conc. HCl–Mn and Mn_T were also linearly related ($r = 0.96$). The acid-extractable manganese probably originates predominantly from secondary manganese compounds, of non-detrital origin, in association with iron compounds (see below).

Sample H13 is characterized by very high Fe_T and Mn_T contents (7.7% and 2800 $mg \cdot kg^{-1}$, respectively). The sediment at this location consists of fine sand coated with black material overlying glacial till. This sample appears to be from an area where iron and manganese oxides are accumulating at the sediment water interface. Similar areas in Lake Ontario have been described by Cronan and Thomas (1972).

TABLE 3. Degree of association of all parameters, as measured by simple correlation coefficients. Upper right half: 43 samples (W40, W42, C15, H13, and M27 always excluded) unless otherwise indicated. Lower left half: all samples unless otherwise indicated.

	Apatite-P	NAI-P	Organic P	Inorganic P	Total P	Fe_T	Conc. HCl–Fe	Mn_T	Conc. HCl–Mn	Organic C	Mean ϕ	Sand	Silt	Clay
Apatite-P		−0.36	−0.17	0.38	0.23	−0.26	−0.31[a]	0.02	−0.03[a]	−0.14	0.15	−0.23	0.52	−0.09
NAI-P	−0.23		0.80	0.72	0.81	0.86	0.83[a]	0.55	0.52[a]	0.76	0.61	−0.54	0.08	0.75
Organic P	−0.22	0.51		0.65	0.83	0.79	0.76[a]	0.36	0.37[a]	0.95	0.74	−0.71	0.33	0.81
Inorganic P	0.50	0.73	0.32		0.97	0.66	0.58[a]	0.60	0.54[a]	0.64	0.70	−0.68	0.44	0.68
Total P	0.41	0.77	0.50	0.98		0.76	0.69[a]	0.57	0.53[a]	0.80	0.77	−0.75	0.44	0.78
Fe_T	0.37	0.50	0.32	0.71	0.71		0.96[a]	0.59	0.56[a]	0.77	0.61	−0.56	0.10	0.76
Conc. HCl–Fe	−0.19[b]	0.67[b]	0.43[b]	0.47[b]	0.52[b]	0.80[b]		0.48[a]	0.47[a]	0.74[a]	0.56[a]	−0.52[a]	0.09[a]	0.70[a]
Mn_T	0.44	0.28	−0.01	0.56	0.51	0.78	0.52[b]		0.92[a]	0.28	0.31	−0.27	−0.04	0.43
Conc. HCl–Mn	−0.07[b]	0.46[b]	0.10[b]	0.36[b]	0.35[b]	0.57[b]	0.66[b]	0.80[b]		0.30[a]	0.30[a]	−0.30[a]	−0.01[a]	0.42[a]
Organic C	−0.25	0.64	0.88	0.43	0.57	0.42	0.58[b]	−0.01	0.13[b]		0.75	−0.75	0.42	0.79
Mean ϕ	−0.03	0.33	0.73	0.31	0.43	0.28	0.32[b]	−0.01	0.02[b]	0.73		−0.96	0.64	0.92
Sand	0.12	−0.32	−0.73	−0.23	−0.36	−0.16	−0.29[b]	0.11	−0.01[b]	−0.74	−0.95		−0.79	−0.85
Silt	0.03	0.15	0.38	0.16	0.23	−0.08	0.05[b]	−0.24	−0.12[b]	0.48	0.67	−0.81		0.37
Clay	−0.20	0.37	0.81	0.22	0.37	0.30	0.40[b]	0.02	0.11[b]	0.75	0.92	−0.86	0.42	

[a]42 samples (H27 also omitted).
[b]47 samples (H27 omitted).

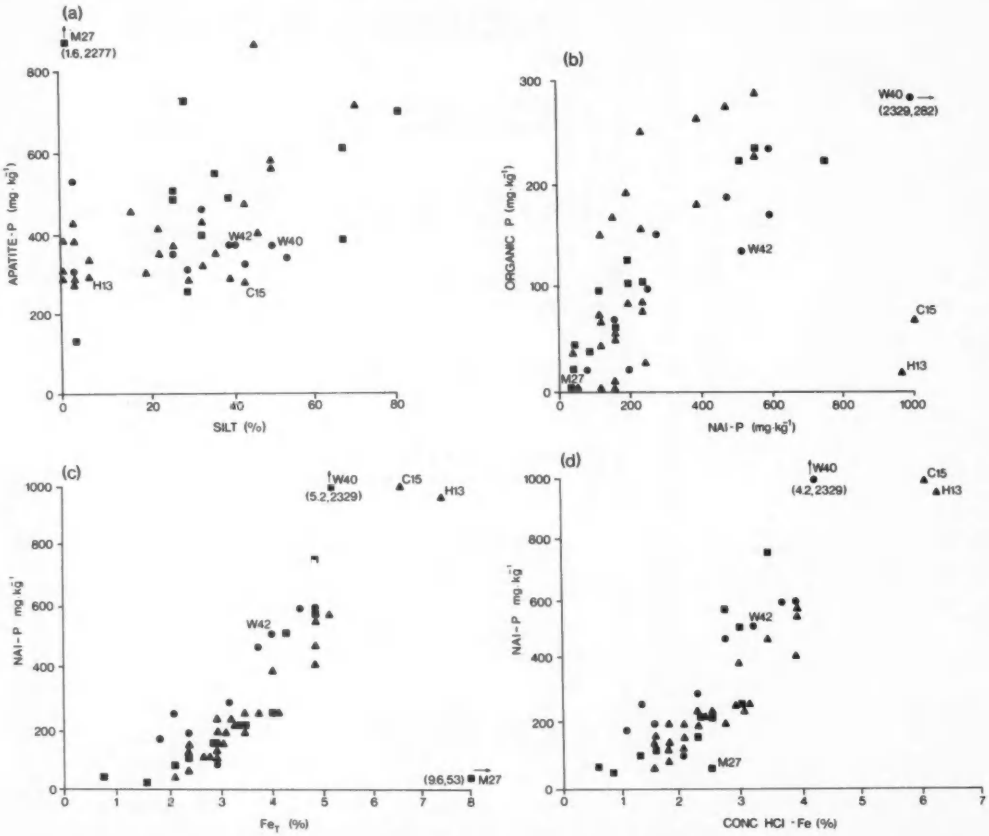

FIG. 4. Scatter diagrams of (a) apatite-P against silt content; (b) organic P against NAI-P; and NAI-P against (c) Fe_T and (d) conc. HCl–Fe. The symbols ●, ▲ and ■ denote samples from the Western, Central, and Eastern basins, respectively.

PHOSPHORUS FORMS AND OTHER SEDIMENT PARAMETERS

Apatite-P associated with only one other parameter, namely silt content. In this comparison, if one sample (M27) was excluded, r was equal to 0.50 (Fig. 4 and Table 3). Silt-sized and clay-sized apatite of subangular morphology (indicating a detrital origin) has been observed by electron microscope in sediment from stations E11 and H21 (P. W. Kingston personal communication 1973). Apatite is perhaps also concentrated to some extent in the fine sand fraction of these materials, but not in the coarse sand fraction. Sample M27, with an apatite-P content exceeding 2000 $mg \cdot kg^{-1}$, was well sorted sediment, with a mean grain size of 3.1 ϕ but a silt content of only 1.6%. In another coarser but also well sorted sample, I28, containing over 90% sand and a mean grain size of 1.0 ϕ, apatite-P was only 144 $mg \cdot kg^{-1}$.

The NAI-P parameter, though not related to apatite-P, was significantly correlated to most parameters examined in this study, including organic P. By excluding three samples (W40, H13, and C15) the correlation coefficient relating these two measurements was increased to 0.80 (Fig. 3b). The NAI-P fraction was also well correlated with the Fe and Mn parameters and with organic C and clay content, provided that in each case, one, two or three sediments out of a group of four (W40, H13, C15, and M27) were excluded (Fig. 4 and 5). The C15 sample is discussed below. The comparison of NAI-P with conc. HCl–Fe (Fig. 4d) revealed that only sample W40 was "abnormal." Without this sample r was

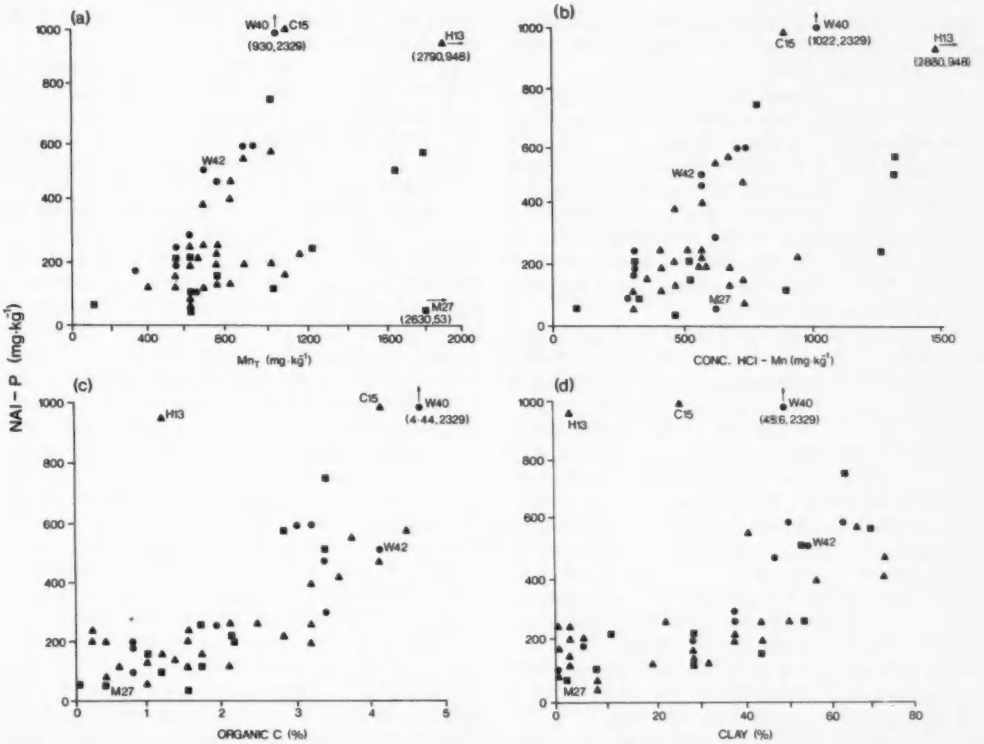

FIG. 5. Scatter diagrams of NAI-P against (a) Mn_T, (b) conc. HCl–Mn, (c) organic C, and (d) clay. Symbols as in Fig. 4.

equal to 0.89. The NAI-P/Fe_T relation was also close, with r equal to 0.91 provided two samples (W40 and M27) were eliminated. The relation of NAI-P with organic C and with clay appears to be curvilinear (Fig. 5c and 5d).

In this study no variable appears to be directly dependent on only one other variable, and analytical error accounts for only a small part of the total deviation of observed values from a linear relationship. Consequently, strictly objective criteria which can be used for calculating a single line of best fit for each pair of variables (Kendall and Stuart 1973) do not seem to exist. Nevertheless, Fig. 4c and 4d suggest that the relations between NAI-P and both Fe_T and conc. HCl–Fe can both be approximated by equations of the form $y = ax - |b|$. The amounts of total and acid-extractable iron in the sediments are so much greater than the amounts of NAIP present that the former can perhaps be justifiably regarded as "independent" variables with respect to NAIP. (Reversing these assumptions has little effect on the conclusions.) In that case, with samples W40, H27, and M27 excluded, conventional linear regression analysis yields the following equations:

$$(\text{NAI-P}) = (162 \pm 11)(\text{Fe}_T) - (305 \pm 43)(r = 0.91) \ldots (1)$$

and

$$(\text{NAI-P}) = (184 \pm 13)(\text{conc. HCl–Fe}) - (227 \pm 41)(r = 0.90) \ldots (2)$$

(units as in Table 1). The slopes of the lines do not differ significantly, but the intercepts differ significantly from zero, and by about two standard deviations from each other.

These equations, but with x and y reversed, are displayed in Fig. 6. The diagram suggests that the iron not extracted by concentrated hydrochloric acid is not associated with inorganic phosphate. According to Table 2, this iron averages about 1.0% of total sediment dry weight. A major part of this iron is probably accounted for by minerals

such as amphiboles, pyroxenes, ilmenite and magnetite, all of which have been observed in sediments near Point Pelee (Coakley 1972).

Figure 6 also suggests that acid-extractable iron can be divided into two components, one of these ("phosphate-reactive iron") being associated with NAIP; however, this study does not supply direct experimental evidence on the relative magnitudes of these two components. Line OA, representing one possible partition of the two components, is thought to represent the situation in the oxidised microzones but not in the reduced zones of Lake Erie sediments (see below).

The unusually high NAIP:iron ratio in sample W40, taken from the mouth of the Detroit River, is undoubtedly a consequence of the large amounts of phosphorus, of municipal or industrial origin, entering this area of Lake Erie via the river from the Detroit metropolitan area. The sample was also characterized by high chromium and cadmium values (R. L. Thomas personal communication 1975): it may thus be greatly affected by industrial discharges, e.g. from electroplating cells.

The NAI-P values were also associated with the Mn parameters, but the degree of association, as measured by correlation coefficients, was considerably less close than with Fe (Table 3). In the comparison with conc. HCl–Mn, only sample W40 was "abnormal," while both W40 and M27 were "abnormal" in the comparison with Mn_T (Fig. 5a and 5b). In view of the close association also found between, for example, conc. HCl–Fe and conc. HCl–Mn (Table 3), it appears that the association between NAIP and manganese is not a direct one but results from the association of both factors with iron in the sediments. This is illustrated by partial and multiple correlation coefficient analysis for the three parameters NAIP, conc. HCl–Fe and conc. HCl–Mn, sample W40 excluded (Table 4).

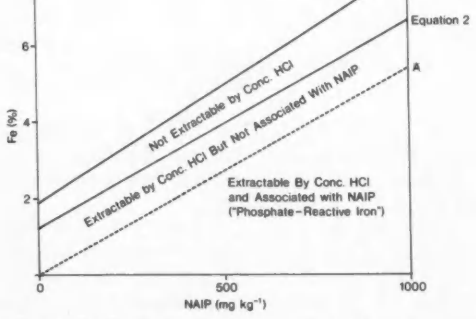

FIG. 6. Idealized representation of the forms of iron and the iron–NAIP relationship in Lake Erie sediments.

In addition to the Fe and Mn parameters, NAI-P was also closely related to organic C and clay content, with correlation coefficients of 0.79 and 0.77, respectively, provided three samples (W40, H13, and C15) were excluded. Multiple and partial correlation coefficient analysis (Table 4), however, suggests that NAI-P was associated directly with conc. HCl–Fe but only indirectly with organic C or clay content. It is well known that the Fe, Mn, and P contents of sediments from the same lake are often closely associated: for example, Serruya (1971) found this to be so in Lake Kinneret. The findings here suggest that even closer associations may be uncovered if, instead of total values, measures of the more readily extractable forms of the elements are used.

It is also interesting that Fig. 4c and 4d suggest that the relations between NAI-P, on the one hand, and organic C or clay content, on the other, are slightly curvilinear upwards. This suggests that sediments of high clay content have more NAIP per unit of clay than coarser sediments. This is explicable if (a) finer clay particles occupy a greater proportion of the total clay fraction in sediments of high clay content and if (b) the finer clay particles, by virtue of their greater surface area to volume ratio, are associated with greater amounts of organic matter, iron compounds and NAIP than coarser clay particles. This hypothesis is currently being tested.

The organic P values were also significantly correlated with all the P parameters except apatite-P, and with Fe_T, conc. HCl–Fe, organic C and clay content (Table 3). The closest correlation when all samples were included was with organic C ($r = 0.88$). The exclusion of only two samples (W42 and C15), however, increased the correlation coefficient to 0.95 (Fig. 7a). The degree of association between these two values in the Lake Erie samples is closer than was observed in a group of Wisconsin lake sediments (Williams et al. 1971), possibly because all samples in this study were taken from the same lake.

If organic C is formally designated the independent variable, the regression equation relating these parameters (W42 and C15 excluded) is:

$$(\text{organic P}) = (80 \pm 4)\ (\text{organic C}) + (-36 \pm 8) \quad \ldots (3)$$

(units as in Table 1). This suggests that the amount of organic phosphorus in these sediments is almost directly proportional to the amount of organic carbon present. The organic C:organic P ratio, based on the slope of equation (3), is about 124:1. The same ratio, calculated from the

TABLE 4. Degree of association between five groups of three parameters, as measured by simple, multiple, and partial correlation coefficients.

Parameter			Samples excluded	Correlation coefficients								
				Simple			Multiple			Partial		
1	2	3		r_{12}	r_{23}	r_{31}	$R_{1.23}$	$R_{2.31}$	$R_{3.12}$	$r_{12.3}$	$r_{23.1}$	$r_{31.2}$
NAI-P	Conc. HCl–Fe	Conc. HCl–Mn	W40	0.89	0.65	0.61	0.89	0.90	0.66	0.82	0.28	0.10
NAI-P	Conc. HCl–Fe	Organic C	C15, H13, W40	0.82	0.73	0.78	0.86	0.83	0.80	0.58	0.26	0.47
NAI-P	Conc. HCl–Fe	Clay	C15, H13, W40	0.82	0.69	0.77	0.86	0.82	0.77	0.62	0.17	0.48
Organic P	Organic C	Clay	C15, W42	0.95	0.78	0.82	0.96	0.95	0.82	0.87	0.01	0.39

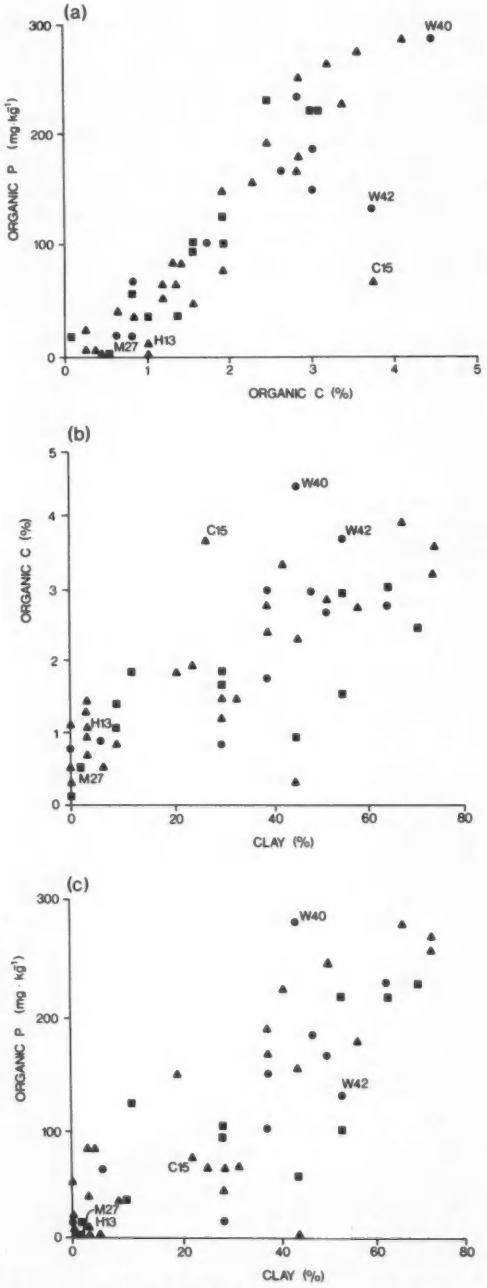

FIG. 7. Scatter diagrams of (a) organic P against organic C; (b) organic C against clay; and (c) organic P against clay. Symbols as in Fig. 4.

mean value of the parameters for the 46 "normal" sediments, is 165:1. These values are somewhat greater than the value of 100:1 which is regarded as a typical value of the organic C:organic P ratio in moderately fertile terrestrial soils (Barrow 1961).

With all samples included, both organic C and organic P were quite closely related to clay content (Fig. 7b and 7c). Table 4 indicates that organic carbon and not clay was the main determinant of the organic phosphorus content of these sediments; however, it is interesting to note that samples W42 and C15 were not "abnormal" in the clay–organic P comparison.

Inspection of Table 3 indicates that organic P was fairly closely related to the same parameters that NAI-P was related to, not surprisingly in view of the association between NAI-P and organic P (see above). In the comparisons of organic P with Fe_T and conc. HCl–Fe sample W40 was not "abnormal." Possibly large amounts of inorganic, but not organic, phosphorus have been introduced into the lake in the vicinity of this sample, again supporting the idea that much of this phosphorus is of industrial origin. By excluding H13 and M27 for the organic P/conc. HCl–Fe comparison r was increased from 0.44 to 0.78. As with clay, the association of organic phosphorus with iron in these sediments does not appear to be direct but is the consequence of the relation of both organic phosphorus and iron to the organic carbon and clay contents of the sediments.

The organic C:organic P ratio by weight of sample C15 was 600. The anomalously low organic P value of this sample may be connected in some way with the fact that this sample was taken from a location within the area used as a dumping ground for spoils dredged from Cleveland harbor. The relation between NAI-P and Fe parameters of this sample was "normal." Sample W42, taken from close to the mouth of the Detroit River, was also low in organic P relative to organic C, but no other unusual features were observed.

Because the inorganic phosphorus of the sediments is partly composed of NAIP and because the total phosphorus is composed of both NAIP and organic phosphorus, the parameters inorganic P and total P were all significantly related, at the 0.1% level, with the same parameters to which NAI-P was significantly related. The r values for comparisons involving inorganic P and total P, however, were lower than those for NAI-P, due to the contribution of apatite-P (Table 3).

The sand content was negatively correlated with all parameters examined in this study. The bulk of the sand fraction is composed of quartz and

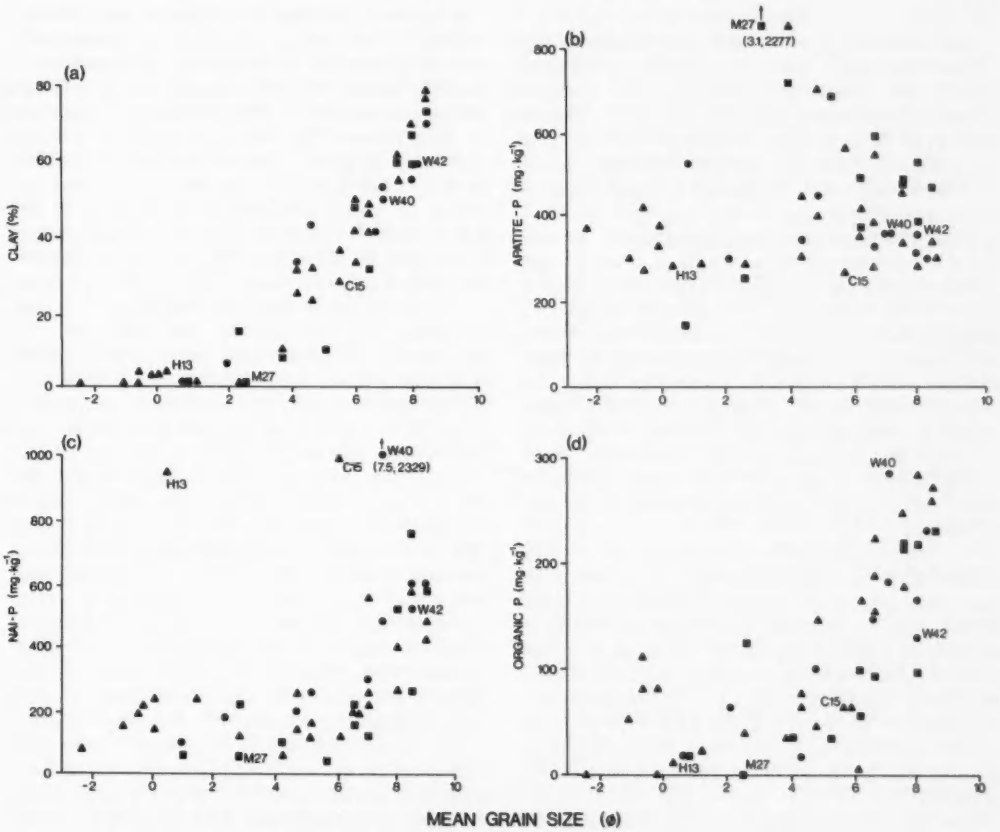

FIG. 8. Scatter diagrams of mean grain size expressed in ϕ units (mean ϕ) against (a) clay, (b) apatite-P, (c) NAI-P, and (d) organic P. Symbols as in Fig. 4.

feldspar of detrital origin and essentially inert chemical behavior. Apart from the correlation with apatite-P, silt was similar to clay in its relation to other parameters, but the correlations were very much less close, indicating that the silt fraction possesses many of the properties of the clay fraction, but to a much lesser degree.

The clay content and the mean grain size of the samples, expressed in ϕ units ("mean ϕ"), are both measures of how fine-grained the samples are, and in fact the two parameters were quite closely related in this study (Fig. 8a). The correlation coefficient of 0.92 underestimates the closeness of the relation, which is strongly curvilinear. Figure 8b shows that the five samples of highest apatite-P content had mean ϕ values ranging from 3.1 to 5.6. This range covers the very fine sand and the coarse silt fractions, according to the Wentworth scale. The relation between NAI-P and mean ϕ is also generally curvilinear. Most samples can be divided into two groups. Those whose mean ϕ value was 6 or less (22 samples) had NAI-P values which ranged from 38 to 241 and averaged 140 ± 69 mg·kg^{-1}. The NAI-P values of the remaining samples (23 in all) were positively correlated with mean ϕ ($r = 0.78$), and the regression equation relating these quantities was:

$$\text{(NAI-P)} = (167 \pm 29)\ \text{(mean } \phi) + (-920 \pm 230) \quad \ldots (4)$$

The relation between mean ϕ and organic P is also curvilinear (Fig. 8d). Lam and Jaquet (1976) have utilized these relationships in their discussion of regeneration of phosphate from Lake Erie sediments.

Discussion

The locations from which the sediments examined here were taken vary greatly in distance from major sources of phosphorus for the lake (especially the large cities surrounding it). Despite this, with the exception of sample W40, there is little evidence to suggest declining sediment phosphorus values with increasing distance from a major phosphorus source. For example, the fine-grained sediments of the Eastern basin do not have lower phosphorus values than those of the Western basin on average. Instead, the values of two of the categories of sediment phosphorus recognized here are closely associated with several chemical parameters of the sediments. In particular, NAIP is closely associated with iron content, while organic phosphorus is closely associated with organic carbon content. In addition, the iron and organic carbon contents are associated, albeit less closely, with each other and with the clay and manganese contents of the sediments.

The close association between organic P and organic C for nearly all samples in this study suggests that organic phosphorus is an integral part of the organic matter of these sediments. A close association between organic C and total N (most of it organic) has been observed in Great Lakes sediments by Kemp (1971). This suggests that the organic matter of Lake Erie sediments is similar in character throughout the lake, differing only in total amount present from one location to aonther.

The associations between clay content, organic C, and the Fe and Mn parameters, though less close than those between organic C and organic P or between conc. HCl–Fe and NAIP, were nevertheless highly significant (see above). To account for these associations in the oxidized microzone, it is assumed that the controlling factor is the component present in greatest amount by weight, namely the amount of clay-sized aluminosilicate particles.

The major components of the clay fraction of Lake Erie sediments are the phyllosilicate minerals illite and chlorite (Lewis 1966). Both minerals carry permanent negative charges, in contrast to iron oxides. During the weathering of terrestrial soils, clay minerals become coated with iron and other oxides (Carroll 1958). In terrestrial soils intimate associations of clay minerals and organic matter also develop, resulting in the partial stabilization of the organic matter against further decomposition (Bremner 1951).

It is therefore concluded that the clay fraction in the oxidized microzone of Lake Erie sediments is an intimate mixture of phyllosilicate minerals, hydrated ferric oxide, and organic matter, with each of these three major components present in the clay fraction in approximately equal amounts throughout the lake. The relative constancy of the proportions of the three components is somewhat surprising, in view of the fact that most production of organic matter in Lake Erie waters occurs at the western end of the lake, far removed from the northern shore and Eastern basin of the lake, where most of the erosion and deposition of fine-grained material occurs. This problem is discussed further by Kemp et al. (1976) and by Burns et al. (1976). The description of the clay fraction given here resembles the postulate of Mattson et al. (1950) that terrestrial soils contain a "phospho-hydroxy-silico-humate complex," of gel-like nature. This complex was thought to play a major role in the chemistry of many anions in soils. It is postulated that to some extent the phyllo-silicate–organic matter–iron association is preserved in the reduced zone, except that the iron compound is no longer hydrated ferric oxide and minor components, such as manganese and orthophosphate, occur in other forms.

The relation between iron and NAIP in these samples remains to be accounted for. Most samples were mixtures of material from the oxidized microzone and the topmost part of the reduced zone. For reasons that are discussed elsewhere (Williams et al. 1976), however, the differences between the NAIP contents of the two types of material comprising a given sample are not great in comparison with the variation in NAIP from one sample to the next; furthermore, there appears to be little vertical migration of orthophosphate from reduced zone to oxidized microzone in Lake Erie sediments (Williams et al. 1976). Consequently, it is assumed that statistical relationships between NAI-P and Fe_T or conc. HCl–Fe of the type discussed above apply to materials from the oxidized microzone and the reduced zone alike.

The association of iron with NAIP in the oxidized microzone is considered first. The brown or reddish color of the microzone indicates the presence of hydrated ferric oxide, whose formula can be written as $Fe(OH)_3$ or $Fe_2O_3 \cdot nH_2O$ (Mortimer 1941, 1942). Amorphous hydrated ferric oxide has an iso-electric point (IEP) of 8.5 (van Schuylenborgh and Sänger 1949) and, at pH values below the IEP, can remove as much as 5% of its own weight of phosphorus in orthophosphate form from solution (Stamm and Kohlschütter 1965). The formula of the resulting product can be written as $Fe_x(OH)_{3(x-y)}(PO_4)_y \cdot zH_2O$: it is termed

here a "ferric oxide–orthophosphate complex." There is as yet no general agreement as to whether this and similar complexes are essentially homogeneous or whether they consist of an intimate mixture of metal phosphate and metal oxide domains (Syers and Williams 1976).

The pH values of Lake Erie surficial sediments lie mostly in the range 7.3–7.5 (Kemp 1971), below the IEP of amorphous hydrated ferric oxide. Consequently, it is proposed here that the association between iron and NAIP in these sediments arises ultimately from the deposition or formation at the oxidized microzone of material similar in composition to the complex proposed above. The material may have incorporated in its structure other cations and anions, and may well be the same as, or similar to, the complex material, containing Al, Fe, Si, Ca, and P, that was detected by electron microprobe in soils and lake sediments by Sawhney (1973). An estimate of the ratio of iron to phosphorus in the complex in the oxidized microzone of Lake Erie sediments is obtained from the slope of equation (2). The value of this estimate is 54 by weight, equivalent to an atomic ratio of 31.

Under actively reducing conditions, however, the complex is destroyed, with liberation of orthophosphate ions. If the hypolimnion is anoxic the released phosphate will diffuse into this water mass, but reoxygenation of the hypolimnion will cause the complex to re-form (Mortimer 1941, 1942). Ultimately the complex is permanently destroyed when the sediment layer of which it is part enters the reduced zone due to continuing sediment accumulation. The reaction is accompanied by liberation of orthophosphate ions into the sediment pore waters (Nriagu and Dell 1974). The same change liberates manganese in soluble Mn^{II} form from other sediment components. This manganese migrates upwards into the oxidized microzone, where it is precipitated from solution, resulting in higher Mn contents in the microzone than in the reduced material below (Robbins and Callender 1975; Kemp et al. 1976). Vertical migration of orthophosphate due to the same cause is much less marked, however (Williams et al. 1976). Consequently, most of the liberated orthophosphate appears to remain associated with the solid phase even in the reduced zone.

One mode of combination of orthophosphate in the reduced zone of Lake Erie sediments is as vivianite ($Fe_3(PO_4)_2 \cdot 8H_2O$. The presence of this mineral in this lake has been established by direct mineralogical observation and by pore water analysis (Nriagu and Dell 1974). Vivianite is readily soluble in CDB reagent (E. Callender personal communication 1975) and hence is operationally as well as by definition part of the NAIP fraction. However, vivianite is not necessarily the only mode of combination of phosphate with iron in the reduced zone. Other possible modes of combination have been discussed by Williams et al. (1971) and by Patrick and Khalid (1974).

Summary and Conclusions

It is postulated that the phosphorus in Lake Erie sediments can be divided into three categories, namely apatite phosphorus, nonapatite inorganic phosphorus (NAIP), and organic phosphorus. The analytical scheme used here to estimate the three categories in 48 surficial Lake Erie sediments was described in outline. The resulting values were compared with total and acid-extractable iron and manganese, organic carbon, mean grain size and sand, silt and clay contents determined on the same samples. All but five of the samples behaved in a "regular" manner in all comparisons. Atypical features of these five sediments are discussed.

Variation in the concentration of the three phosphorus categories was to a large extent explicable. Most, if not all, apatite phosphorus is of detrital (used in its geological sense) origin, the most important single source being the bluffs lining the northern shore of the Central basin. The coarser sedimented apatite grains tend to be deposited preferentially in nearshore areas, especially west and south of Long Point, and the finer clay-sized materials in offshore depositional areas. Apatite phosphorus was weakly related to only one other variable, silt content.

By contrast, both NAIP and organic phosphorus are present mainly in the fine-grained sediments accumulating in offshore depositional areas. The NAIP content was closely related to the reactive iron content of the sediments, while organic phosphorus content was closely related to organic carbon. Both categories of phosphorus were also related to each other and to the clay and manganese contents of the sediments. In the oxidized microzone, NAIP is thought to be mainly associated with amorphous hydrated ferric oxide. The approximate atomic ratio of iron to phosphorus in the resulting "ferric oxide–orthophosphate complex" is about 54 by weight. In the reduced zone this complex is destroyed, but essentially the same iron:phosphorus ratio is "inherited" by the sum of the products of this diagenetic reaction. Vivianite ($Fe_3(PO_4)_2 \cdot 8H_2O$) and possibly other modes of combination of

orthophosphate with ferrous iron are formed instead.

Organic phosphorus exists as an integral part of sediment organic matter, whose chemical nature perhaps does not vary very much throughout the lake. Phyllosilicate clay minerals, organic matter and ferric oxide, together with associated NAIP and organic phosphorus, appear to exist in intimate association in the oxidized microzone, as in freely-drained terrestrial soils. The ratios of these components remain essentially unaltered in the reduced zone, though new modes of combination of iron are created.

The absence of any trend in sediment phosphorus values from west to east that could be attributed to major phosphorus inputs indicates that, within this lake, the distribution of sedimented phosphorus is determined primarily by other sediment properties, which are in turn determined ultimately by the control on sedimentation exerted by physical factors such as water movement.

Estimates of sediment inorganic phosphorus content were obtained in two ways, one by summing apatite-P and NAI-P values. Both estimates agreed well. Because the two inorganic phosphorus categories showed different behavior, the sum was less closely related to other parameters than the NAI-P values. Similarly, total phosphorus values, obtained by summing the estimates of the three phosphorus categories, were not closely related to other measured variables.

The study also shows that only part of the iron in sediments will react with orthophosphate, the unreactive portion being that associated with primary minerals and possibly chlorite and other forms of iron also. In addition, the hypothesis that sediment phosphorus exists in three different major forms, identifiable by selective dissolution techniques and by the different ways in which they are associated with other sediment components, is partially verified. This indicates that before the behavior of many elements in sediments (including phosphorus, iron, manganese, and perhaps several heavy metals) can be fully described, the chemical modes of combination of these elements must first of all be elucidated.

Acknowledgments

We would like to thank members of the Technical Operations Section, C.C.I.W. and the officers and crew of the C.S.S. *Limnos* for assistance in sampling, and T. Mayer, T. W. Morton, G. Bengert, and staff of the Sedimentology Laboratory for analytical support. We are indebted to J. W. Morse and R. Hallberg for formally reviewing the manuscript and suggesting several improvements, and to N. M. Burns and many other colleagues at C.C.I.W. for helpful and critical discussion throughout the study. We thank Ethel Kerr and Marilyn Stapleton for typing the manuscript, and P. G. Sly, Chief, Process Research Division, for permission to publish.

ANDERSON, G. 1960. Factors affecting the estimation of phosphate esters in soil. J. Sci. Food Agric. 11: 497–503.

BARROW, N. J. 1961. Phosphorus in soil organic matter. Soils Fert. 24: 169–173.

BREMNER, J. M. 1951. A review of recent work on soil organic matter, Part 1. J. Soil Sci. 2: 67–82.

BURNS, N. M., J. D. H. WILLIAMS, A. L. W. KEMP, J.-M. JAQUET, AND D. C. L. LAM. 1976. A phosphorus budget for Lake Erie. J. Fish. Res. Board Can. 33: 564–573.

CARROLL, D. 1958. Role of clay minerals in the transportation of iron. Geochim. Cosmochim. Acta 14: 1–28.

COAKLEY, J. P. 1972. Nearshore sediment studies in western Lake Erie. Proc. Conf. Great Lakes Res. 15: 330–343.

CRONAN, D. S., AND R. L. THOMAS. 1972. Geochemistry of ferromanganese concretions and associated deposits in Lake Ontario. Geol. Soc. Am. Bull. 83: 1493–1502.

INTERNATIONAL JOINT COMMISSION. 1969. pollution of Lake Erie, Lake Ontario and the international section of the St. Lawrence River. Vol. 1 — Summary. (Mimeo) 150 p.

KEMP, A. L. W. 1971. Organic carbon and nitrogen in the surface sediments of Lakes Ontario, Erie and Huron. J. Sediment. Petrol. 41: 537–548.

KEMP, A. L. W., AND C. F. M. LEWIS. 1968. A preliminary investigation of chlorophyll degradation products in the sediments of Lakes Erie and Ontario. Proc. Conf. Great Lakes Res. 11: 206–229.

KEMP, A. L. W., R. L. THOMAS, C. I. DELL, AND J.-M. JAQUET. 1976. Cultural impact on the geochemistry of sediments in Lake Erie. J. Fish. Res. Board Can. 33: 440–462.

KENDALL, M. G., AND STUART, A. 1973. The advanced theory of statistics. Vol. 2: Inference and relationship. 3d ed. Hafner, New York, N.Y. 723 p.

LAM, D. C. L., AND J.-M. JAQUET. 1976. Computations of physical transport and regeneration of phosphorus in Lake Erie, fall 1970. J. Fish. Res. Board Can. 33: 550–563.

LEWIS, C. F. M. 1966. Sedimentation studies of unconsolidated deposits in the Lake Erie basin. Unpublished Ph.D. Thesis. Univ. Toronto, Toronto, Ont. 134 p.

MATTSON, S., E. ALVSAKER, E. KOUTLER-ANDERSSON, E. BARKOFF, AND K. VAHTRAS. 1950. Phosphate relationships of soil and plant. VI. The salt effect on phosphate solubility in pedalfer soils. Lantbrukshoegsk. Ann. 17: 141–160.

MORTIMER, C. H. 1941. The exchange of dissolved substances between mud and water in lakes. J. Ecol. 29: 280–329.

1942. The exchange of dissolved substances between mud and water in lakes. J. Ecol. 30: 147–201.

NRIAGU, J. O. 1972. Stability of vivianite and ion-pair formation in the system $Fe_3(PO_4)_2$-H_3PO_4-H_2O. Geochim. Cosmochim. Acta 36: 459–470.

NRIAGU, J. O., AND C. I. DELL. 1974. Diagenetic formation of iron phosphates in recent lake sediments. Am. Mineral. 59: 934–946.

PATRICK, W. H., JR., AND KHALID, R. A. 1974. Phosphate release and sorption by soils and sediments: effect of aerobic and anaerobic conditions. Science 186: 53–55.

ROBBINS, J. A., and CALLENDER, E. 1975. Diagenesis of manganese in Lake Michigan sediments. Am. J. Sci. 275: 512–533.

ROBERTSON, G. 1958. Determination of phosphate in citric acid extracts. J. Sci. Food Agric. 9: 288–294.

SAWHNEY, B. L. 1973. Electron microprobe analysis of phosphates in soils and sediments. Soil Sci. Soc. Am., Proc. 37: 658–660.

SCHUYLENBORGH, J. VAN, AND A. M. H. SÄNGER. 1949. The electrokinetic behavior of iron and aluminum hydroxides and oxides. Rec. Trav. Chim. Pays-Bas Belg. 68: 999–1010.

SERRUYA, C. 1971. Lake Kinneret: the nutrient chemistry of the sediments. Limnol. Oceanogr. 16: 510–521.

SOMMERS, L. E., R. F. HARRIS, J. D. H. WILLIAMS, D. E. ARMSTRONG, AND J. K. SYERS. 1972. Fractionation of organic phosphorus in lake sediments. Soil Sci. Soc. Am., Proc. 36: 51–54.

STAMM, H.-H., AND KOHLSCHÜTTER, H. W. 1965. Sorption of phosphate ions on iron (III) hydroxide. J. Inorg. Nucl. Chem. 27: 2103–2108.

SYERS, J. K., R. F. HARRIS, AND D. E. ARMSTRONG. 1973. Phosphate chemistry in lake sediments. J. Environ. Qual. 2: 1–14.

SYERS, J. K., AND J. D. H. WILLIAMS. 1976. Sources, amounts and forms of phosphorus and arsenic in soils, *In* J. M. Bremner and G. Chesters [ed.]. Soil chemistry. Marcel Dekker, New York, N.Y. (In press)

THOMAS, R. L., A. L. W. KEMP, AND C. F. M. LEWIS. 1972. Distribution, composition and characteristics of the surficial sediments of Lake Ontario. J. Sediment. Petrol. 42: 66–84.

THOMAS, R. L., J.-M. JAQUET, A. L. W. KEMP, AND C. F. M. LEWIS. 1976. Surficial sediments of Lake Erie. J. Fish. Res. Board Can. 33: 385–403.

WILLIAMS, J. D. H., AND T. MAYER. 1972. Effects of sediment diagenesis and regeneration of phosphorus with special reference to Lakes Erie and Ontario, p. 281–315. *In* H. E. Allen and J. R. Kramer [ed.]. Nutrients in natural waters. Wiley-Interscience, New York, N.Y.

WILLIAMS, J. D. H., T. P. MURPHY, AND T. MAYER. 1976. Rates of accumulation of phosphorus forms in Lake Erie sediments. J. Fish. Res. Board Can. 33: 430–439.

WILLIAMS, J. D. H., J. K. SYERS, S. S. SHUKLA, R. F. HARRIS, AND D. E. ARMSTRONG. 1971. Levels of inorganic and total phosphorus in lake sediments as related to other sediment parameters. Environ. Sci. Technol. 5: 1113–1120.

WILLIAMS, J. D. H., J. K. SYERS, T. W. WALKER, AND R. W. REX. 1970. A comparison of methods for the determination of soil organic phosphorus. Soil Sci. 110: 13–18.

WILLIAMS, J. D. H., AND T. W. WALKER. 1969. Fractionation of phosphate in a maturity sequence of New Zealand basaltic soil profiles: 2. Soil Sci. 107: 213–219.

Rates of Accumulation of Phosphorus Forms in Lake Erie Sediments

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WILLIAMS, J. D. H., T. P. MURPHY, AND T. MAYER. 1976. Rates of accumulation of phosphorus forms in Lake Erie sediments. J. Fish. Res. Board Can. 33: 430–439.

Six Lake Erie sediment cores from locations of widely different sedimentation rates show that rate of input and sedimentation of apatite phosphorus at a given locality has been approximately constant during the last few hundred years relative to total sedimentation rate at the same locality. Apatite is not a significant source of soluble phosphorus for Lake Erie. By contrast, rate of sedimentation of nonapatite inorganic phosphorus (NAIP) and organic phosphorus has steadily increased in accordance with increased loadings in recent years of the source material, namely phosphorus of anthropogenic origin. The rate of sedimentation of these two forms of phosphorus at a given locality varied greatly within the lake, and was greatest where the most fine-grained sediments were accumulating. It was concluded that the sediments of Lake Erie contain sufficient orthophosphate-retaining sites to prevent major regeneration of phosphate under any conceivable condition provided the overlying water remains oxic.

WILLIAMS, J. D. H., T. P. MURPHY, AND T. MAYER. 1976. Rates of accumulation of phosphorus forms in Lake Erie sediments. J. Fish. Res. Board Can. 33: 430–439.

Six carottes de sédiment du lac Erié provenant de régions à taux de sédimentation très différents indiquent que les apports et taux de sédimentation du phosphore sous forme d'apatite à un endroit donné ont été à peu près constants au cours des quelques derniers siècles par rapport au taux de sédimentation totale au même endroit. L'apatite n'est pas une source importante de phosphore soluble pour le lac Erié. Par contre, le taux de sédimentation du phosphore inorganique non-apatite (PINA) et du phosphore organique a augmenté régulièrement avec l'augmentation des charges, en ces dernières années, du matériel d'origine, soit le phosphore de source anthropogénique. Le taux de sédimentation de ces deux formes de phosphore à un endroit donné varie grandement dans le lac et est à son macimum là où les sédiments les plus fins s'accumulent. Nous en concluons que les sédiments du lac Erié renferment suffisamment de sites de rétention d'orthophosphate pour prevenir une importante régénération du phosphate, quelles que soient les conditions que l'on puisse concevoir, pourvu que l'eau qui recouvre ces sédiments ne manque pas d'oxygène.

Accepted November 17, 1975 Accepté le 17 novembre 1975

A concurrent study (Kemp et al. 1976) discusses rates of sedimentation of phosphorus of natural and anthropogenic origin in Lake Erie from precolonial times (i.e. prior to 1850) to the present. This study discriminated between sedimented phosphorus of natural and anthropogenic origin, but did not indicate in what forms this phosphorus was accumulating. Another study (Williams et al. 1976) categorized sediment phosphorus in Lake Erie into three forms, namely apatite phosphorus, nonapatite inorganic phosphorus (NAIP), and organic phosphorus. The behavior of these three categories in the surficial sediments proved to be to a considerable extent explicable; however, that report gave no information on how the amounts of the different forms varied with depth in the sediment column. This subject, and the information it provides on sedimentation of phosphorus of both natural and anthropogenic origin in the lake during recent years, is considered here.

Printed in Canada (J3996)
Imprimé au Canada (J3996)

Materials and Methods

Sediment cores were taken from six stations spread throughout the length of Lake Erie (Table 1 and Fig. 1). Cores from these stations have been described in detail by Kemp et al. (1976). These authors identified the 1935 and 1850 horizons of these cores by pollen analysis. The rate of sedimentation at these six localities varies in the order 1, $6 > 3 > 2 > 5, 4$ (Table 1). Locations 1 and 3 are at or close to the positions of maximum rate of sedimentation in the Western and Central basins, respectively (Kemp et al. 1976). Locations 4 and 5 are close to the edge of the Central basin sediment depositional area (Thomas et al. 1976). Location 6 represents the region of

TABLE 1. Sampling locations in Lake Erie.

Station		Mean annual accumulation of sediment[a] (*mm*)
No.	Designation[a]	
1	U42	7.6
2	E11	2.8
3	G16	5.1
4	H21	0.8
5	L19	0.8
6	N34	3.3

[a]From Kemp et al. (1976).

FIG. 1. Location of sampling stations.

sediment accumulation in the Eastern basin, while 2 differs from 3 mainly in lower sedimentation rate.

At each location, material down to 20 cm below the sediment–water interface was recovered by scuba divers, who inserted six 50-cm lengths of 7.3 cm diameter cellulose acetate butyrate tubing into the sediments by hand. Six cores were taken from each site and subdivided into 1-cm sections down to 2 cm below the sediment–water interface and 2-cm sections thereafter down to 20 cm. A vertical extrusion technique developed at the Canada Centre for Inland Waters was used for subdividing the cores and ensured minimal disturbance of the highly fluid sediment–water interface. The cores from a given site provided six samples of the material at a given depth below the sediment–water interface and, to obtain sufficient material for many different types of geochemical analysis (most of which are not reported here), the six separate samples were pooled to create one composite sample. This is the material analyzed here. Sediment from depths below 20 cm was recovered by a Benthos® gravity corer (Benthos Inc., Edgerton Drive, North Falmouth, Mass. 02556, USA) and the core subdivided into 5-cm sections. The cores from station 5 were recovered in September 1971 and the remainder in June 1971. All samples were freeze-dried and gently crushed in a pestle and mortar to a uniformly fine composition suitable for subsampling for chemical analysis. The resulting composite cores are identified by station number. One further core from each station was subdivided down to 20 cm into 1- and 2-cm sections as above, and each section weighed after freeze-drying. Assuming a uniform rate of sedimentation of total solids in the period 1935 to present, the weights were used to estimate the time during which the material in each core section was deposited at the sediment–water interface.

Much of the north shore of Lake Erie is composed of bluffs up to 50 m high, which are rapidly eroding, with average rates of recession of up to 3 m per year having been recorded in some localities (Gelinas and Quigley 1973). Kemp et al. (1976) have shown that the bluffs contribute most of the sediment entering the Central and Eastern basins of Lake Erie, particularly in the region around Long Point. The bluffs were sampled at 12 locations along the north shore, ranging from west of Leamington to east of Long Point. These locations, numbered 7 to 18 inclusive in Fig. 1, are the same as the locations similarly numbered in Fig. 1 of Kemp et al. (1976). At most locations, samples were taken from more than one position in the bluff and pooled. Samples taken from the top and bottom of the bluffs at locations 8 and 18 were analyzed separately.

Rates of sedimentation at locations 1 to 6 inclusive were determined by palynological analysis, using contents of *Ambrosia* and *Castanea* pollen grains as horizon markers (Kemp et al. 1976). Particle-size distribution analysis of the sediments was performed by wet sieving and Sedigraph analysis (Duncan and Rukavina 1972), and the calculation from these data of the mean particle size of the sediments expressed in ϕ units ("mean ϕ") was carried out by standard sedimentological methods (Thomas et al. 1972). The higher the value of mean ϕ, the more fine grained the sediment. The procedures used to estimate the content of apatite phosphorus, nonapatite inorganic phosphorus (NAIP), and organic phosphorus in this study were essentially the same as were used for surficial sediments from Lake Erie (Williams et al. 1976). In this study, however, the value of NAI-P was obtained from the amount of orthophosphate in the citrate-dithionite-bicarbonate (CDB) extract alone, and the amount of orthophosphate in the subsequent NaOH extract was not determined. Table 2 of Williams et al. (1976) shows that orthophosphate extracted by the NaOH treatment averaged only about 4% of that extracted by the preceding CDB treatment; consequently, the change in procedure had only a slight effect on the estimated values of NAIP. Estimates of the total phosphorus and total inorganic phosphorus contents of the samples, denoted by the expressions total P and inorganic P, were obtained as described elsewhere (Williams et al. 1976). In addition, estimates of the "nonapatite phosphorus" (NAP) content of the sediments are discussed here. These estimates, denoted by the expression "NA-P," were obtained by summing the values of NAI-P and organic P for a given material.

TABLE 2. Phosphorus fractions in bluff samples from the north shore of Lake Erie.

Location	Position on bluff[a]	Apatite-P	NAI-P	Organic P	Total P
		($mg \cdot kg^{-1}$ *dry sediment*)			
7	A, B	408	12	4	424
8	A	486	12	8	506
8	B	492	4	27	523
9	A, B, C	449	12	0	461
10	A, B, C	550	19	8	577
11	A, B	567	19	18	604
12	A, B, C	560	19	4	583
13	A, B	573	23	0	596
14	A, B	577	23	15	615
15	A, B, C	594	23	36	653
16	A	587	23	14	624
17	A	614	23	17	654
18	A, B	583	27	9	619
18	C	597	19	6	622
Avg		545±61	18±6	12±10	576±71

[a]A, B, and C denote sampling positions at the top, middle and bottom of the bluffs, respectively. (A, B) and (A, B, C) denote composite samples obtained by pooling material from the indicated positions in the bluff.

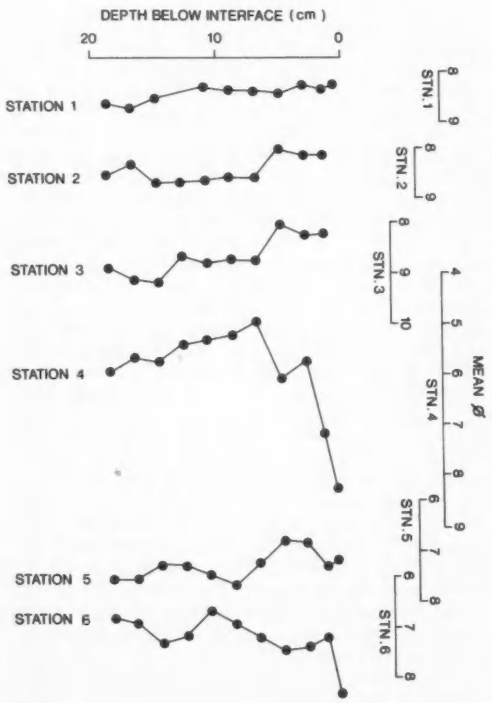

FIG. 2. Variation in mean grain size, expressed in ϕ units, with depth below the sediment–water interface in cores from stations 1 to 6.

Results

PARTICLE-SIZE DISTRIBUTION IN THE LAKE ERIE CORES

The high degree of uniformity during recent times in the composition of the material sedimented at each location (except station 4) is shown by the manner in which mean ϕ varies with depth below the sediment–water interface (Fig. 2). Core 3 contained the finest grained sediments, closely followed by cores 2 and 1. The coarsest sediments were found in core 4, and this was also the only core where sand-size particles (<4 ϕ) comprised more than 1% of sediment dry weight. The standard deviation of the mean ϕ values of all samples for a given core ranged from 0.21 ϕ (core 1) to 0.44 ϕ (core 6), excepting core 4 for which the value was 0.96 ϕ. The enrichment of the top 6 cm of core 4 with fine-grained materials may be due to a recent change in conditions of sedimentation at this locality. Apart from this, sedimentation conditions appear to have varied little at the remaining localities during the time periods spanned by the top 20 cm of these cores.

FORMS OF PHOSPHORUS IN THE BLUFF SAMPLES

Kemp et al. (1976) have estimated that erosion of the shoreline bluffs lining the northern shore of the lake results in a contribution of 23.4 metric tons to the fine-grained sediment loading of the

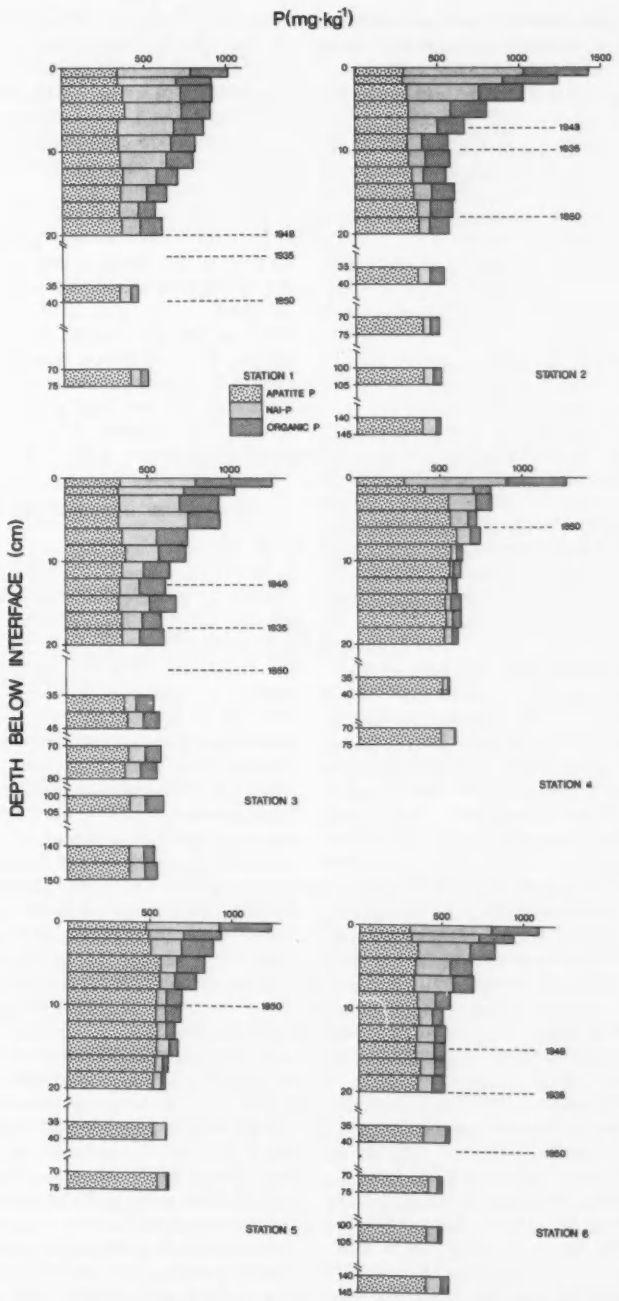

FIG. 3. Profiles of apatite-P, NAI-P, and organic P at the six sediment coring stations.

TABLE 3. Summary of apatite-P values for Lake Erie cores.

Station no.	Depth below interface (*cm*)	N	Apatite-P (*mg·kg⁻¹ dry sediment*) Range	$\bar{X} \pm$ SD
1	0–20	11	339–384	354±16
	20–75	2	348, 402	
2	0–20	11	296–387	331±31
	35–145	4	378–409	
3	0–20	11	320–369	337±14
	35–150	7	351–381	371±9
4	0–2	2	282, 410	
	2–20	9	525–601	554±23
	35–75	2	510–496	
5	0–20	11	475–569	530±29
	35–75	2	523, 550	
6	0–2	2	303, 317	
	2–20	9	337–374	354±11
	35–145	4	392–429	

lake. This contribution is about 67% of the total loading. Any phosphorus present in these materials will likewise enter the lake. Consequently, it is of interest to establish the nature of the phosphorus in the bluffs.

Table 2 shows that the average total P values of the 14 bluff samples did not vary widely, averaging 576 ± 71 $mg \cdot kg^{-1}$. These shoreline bluffs thus contribute 13,500 metric tons to the phosphorus loading of the lake. Much of this eroded material is sedimented in the Eastern basin (Kemp et al. 1976). These authors estimated the total natural phosphorus loading of this basin as 12,300 metric tons, based on analysis of sediment cores.

A feature of the bluff samples is the very low amounts of organic P present, exceeding 20 $mg \cdot kg^{-1}$ in only two of the 14 samples (Table 2). NAI-P values were also generally rather low, being less than 20 $mg \cdot kg^{-1}$ in over half the samples.

The predominant fraction in all samples was apatite-P, which averaged 545 ± 61 $mg \cdot kg^{-1}$, or about 95% of total P, and ranged from 408 to 614 $mg \cdot kg^{-1}$. There was a tendency for the values to increase from west to east along the lake shore (Table 2). NAI-P and organic P averaged only 3 and 2%, respectively, of total P. Following Williams and Walker (1969), this indicates that the bluffs have been subject to very little aerial weathering since their formation during Holocene times. At two locations (8 and 18) the phosphorus values of samples from the top and bottom of the bluffs were very similar. The low NAI-P values also indicate that, even if all the inorganic phosphorus in these samples had consisted of apatite, a maximum of less than 4% of the apatite phosphorus was extracted by the citrate reagent. This establishes an upper limit on the amount of error, in the estimation of apatite phosphorus in these samples, which could arise from premature extraction of apatite phosphorus by the citrate reagent during the sequential extraction procedure.

Although most of the apatite introduced into Lake Erie is transported into and eventually sedimented in the Eastern basin (Burns et al. 1976) the water quality of this area is appreciably better than in the western half of the lake, where most of the phosphorus of anthropogenic origin enters. The conclusion that phosphorus from this source is largely unavailable for algal growth is supported by the results for sediment materials which were deposited on the sediment–water interface prior to 1850 (see below).

FORMS OF PHOSPHORUS IN THE CORES

Precolonial sedimented material — Inspection of the apatite-P, NAI-P, and organic P values of "precolonial" material sedimented prior to 1850 (Fig. 3) indicates that apatite phosphorus was the predominant form deposited in this period, assuming that the sediment record is not unduly distorted by postdepositional transformations or migration of phosphorus. Reasons for believing these assumptions to be substantially correct in Lake Erie are presented below. Apatite-P averaged 59% of total P in all samples dating prior to 1850 in core 3, 74% of total P in the corresponding samples of cores 1, 2, and 6, and 83% in cores 4 and 5. Apatite-P was appreciably greater, both in absolute concentrations and as a percentage of total P, in cores 4 and 5 than in the remainder. The same variable was essentialy constant below the 1850 depth within a given core, including core 3, where material deposited prior to 1850 was analyzed at seven depths (Table 3).

Within the pre-1850 regions of the same cores the values of NAI-P and organic P were essentially constant for a given core, the exception being a decline in organic P with depth in core 2. The values of NAI-P and organic P were very much less than the corresponding apatite-P values in all these materials, and organic-P was especially low in cores 4 and 5.

These results, although limited to only six stations, indicate a positive relation between NAI-P or organic P and mean ϕ, and a negative relation between apatite-P and mean ϕ. This is the same as the relations found between these parameters in the surficial sediments of the same lake (Wil-

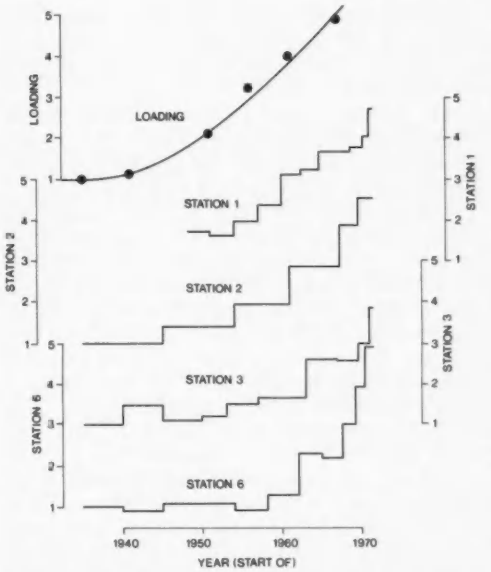

FIG. 4. Comparison of changes in total municipal phosphorus loading and rates of sedimentation of NA-P (=NAI-P + organic P) at stations 1, 2, 3, and 6 during the period 1935 to 1971. All values relative to the corresponding 1935 value.

liams et al 1976). From this it is concluded that essentially the same processes controlled the distribution of apatite phosphorus, NAIP, and organic phosphorus in Lake Erie sediments in the pre-1850 period as operate at present. The major difference is that, based on the profiles of the three forms of phosphorus in Fig. 4, the net accumulation of NAIP and organic phosphorus per unit weight of total sedimented solids prior to 1850 was very much less than today, assuming that total sediment loadings then were not greatly different from the present time.

The results for these six stations suggest that, prior to 1850, apatite phosphorus accounted for between 60 and 80% of the total phosphorus sedimented in the lake. This agrees with an independent estimate obtained by Burns et al. (1976).

Electron microprobe studies on samples from cores 2 and 6 from depths as great as 100 cm below the sediment–water interface have shown that the apatite present was almost entirely, if not solely, of detrital origin, based on the morphology of the grains (P. W. Kingston personal communication 1973). This conclusion is supported by the constancy of apatite-P values with depth below the 1850 horizon in each core. Although the cores can be assumed to cover periods of a few hundreds of years prior to 1850 these time spans are less by one or two orders of magnitude than the core from Lake Ontario that was examined by Williams and Mayer (1972), where diagenetic formation of apatite was postulated. The question of whether diagenetic formation of apatite occurs in Lake Erie sediments is thus unresolved.

The major sources of the detrital apatite entering the lake prior to 1850 were undoubtedly essentially the same as now (Kemp et al. 1976). As fluorapatite ($Ca_{10}(PO_4)_6F_2$) is by far the most common form of primary[1] apatite in nature (Palache et al. 1951) it may be assumed that almost all of the apatite entering Lake Erie is of this variety. The loading of apatite phosphorus into Lake Erie is in excess of 15,000 metric tons (Burns et al. 1976). It is noteworthy that this high input of phosphorus did not render the lake as a whole eutrophic, as evidenced by the historical record and the species composition of the fish population in the 19th century (Sly 1976; Leach and Nepszy 1976). If the same mass of a conservative substance, such as chloride ion, were introduced uniformly into the lake the resultant increase in concentration would be about 0.09 $mg \cdot liter^{-1}$.

The reason why detrital apatite appears to contribute so little available phosphorus for algal growth is undoubtedly associated with the relative insolubility of this mineral, plus its removal from suspension in the lake water by sedimentation. Under natural weathering conditions the rate of dissolution of apatite is very slow; for example, it persists in typical noncalcareous soils, with pH values less than 7, for thousands of years before complete dissolution (William et al. 1969). The average time during which apatite introduced into the lake is in contact with lake waters before burial in the sediment column cannot be estimated at present. However, the mineral is of rather high specific gravity (around 3.1) and occurs in Lake Erie predominantly as silt- or fine sand-sized grains (Williams et al. 1976). As sedimentation is rapid in many locations in the lake (Kemp et al. 1976) it is suggested here that the average time of contact between apatite and lake water does not exceed 100 yr. Apatite is stable once incorporated into the sediment column. Consequently, it is proposed here that not more than about 1% of the detrital apatite entering

[1]"primary" is used in its geological sense to describe a mineral "... formed at the same time as the rock enclosing it, by igneous, hydrothermal, or pneumatolytic processes, and that retains its original composition and form." (Gary et al. 1972).

Lake Erie dissolves in the lake waters, i.e. that the contribution of apatite dissolution to the soluble phosphorus loading of the lake does not exceed 200 metric tons. The exact value may be highly dependent on the particle size distribution of the apatite grains and on water composition including pH value.

Postcolonial sedimented material — The factors influencing the concentration of any substance at a given depth in a core profile can be listed as: rate of sedimentation of that substance relative to total sedimentation rate at the time the substance was deposited at the sediment–water interface; diagenetic changes, often accompanied by vertical migration of the substance through sediment pore waters; and mixing of the surface layers of the sediment column due to bioturbation, gas movement, or turbulent water movement. In the absence of mixing and vertical migration the concentration profile records the rate of deposition of the substance relative to the rate of sedimentation of all solid materials. Surface mixing, however, "blurs" this record, while vertical migration may be sufficient to distort the record and introduce large errors into attempts to deduce past loading rates from concentration profiles. The profiles of manganese in Great Lakes sediment columns bear no relation to sedimentation rates, owing to the pronounced tendency for this element to migrate into the oxidized microzone (Robbins and Callender 1975; Kemp et al. 1976).

The broad features of the 1850 to present regions of the six cores may be summarized as follows. The apatite-P values are fairly constant or increase slowly with increasing depth, whereas the NAI-P and organic P values decline sharply.

The apatite-P values of cores 1 and 3 were essentially constant with depth, while those of cores 2 and 6 showed a slight but steady increase (Table 3 and Fig. 3). In core 4 the highest value was in the 6–8-cm section, which was also the section with the lowest value of mean ϕ, and the apatite-P values of the 0–1 and 1–2-cm sections were only 47 and 68% of this. A slight maximum value of apatite-P at 4–6 cm in core 5 is also associated with a slightly lower mean ϕ value at this depth. Both cores therefore show an inverse relationship between apatite-P and mean ϕ, as is found in Lake Erie sediments in general (Williams et al. 1976). Minor changes in sedimentation conditions appear to have occurred at station 4 and, to a lesser extent, at station 5 within the past hundred years or so, and these have slightly affected the apatite phosphorus profiles at these stations; apart from this, however, the essential uniformity of the apatite-P profiles suggests little change in the processes governing apatite sedimentation in this lake throughout the past few hundred years.

The profiles suggest a common source for the phosphorus accumulating as NAIP and organic phosphorus in these cores. The positive association between mean ϕ and both NAI-P and organic P (Williams et al 1976) may at least partially explain the higher P values in the uppermost layers of core 4 (Fig. 2 and 3), but in the remaining cores mean ϕ was virtually constant with depth. The contrast with the apatite-P profiles suggests that most of the NAIP and organic phosphorus in Lake Erie sediments is not of detrital origin. Virtually all the increase in NAI-P and organic P values occurs in material deposited since 1850, indicating that the increase towards the sediment–water interface is due to increased rates of sedimentation of phosphorus of anthropogenic origin in recent years. It is also possible, however, that part at least of the increase is due to postdepositional migration of phosphorus upwards through the sediment pore waters, as a result of diagenetic reactions in the reduced zone of the sediments (Williams et al. 1976).

A representation of the manner in which the anthropogenic phosphorus loading of Lake Erie has changed in recent years is provided by the estimates of Gilbertson et al. (1972) of total municipal phosphorus loading from 1935 to 1971. Their estimates, expressed on a scale in which the 1935 value is unity, are reproduced in Fig. 4. For comparison, changes in the rates of sedimentation of NAP (nonapatite phosphorus) at stations 1, 2, 3, and 6 are included on the same diagram. Because sedimentation rates vary so widely within the lake (Kemp et al. 1976) values for stations 2, 3, and 6 are expressed relative to the 1935 value at the same station. The 1935 value was not available for station 1. According to Gilbertson et al. (1972) the anthropogenic phosphorus loading of Lake Erie in 1948 was about 1.7 times that of the 1935 value. Consequently, estimated values of sedimentation at location 1 relative to 1935 were obtained by dividing the year value by the 1948 value and multiplying by 1.7.

Figure 4 indicates that it is the NAP sedimentation profile of station 2 which most closely matches the loading profile. The close correspondence suggests that migration of phosphorus upwards was not a significant factor at this locality. The sedimentation profile of station 1 rises less steeply with time than the loading profile. Possibly the sedimentation record here is "blurred," due to turbulent mixing of the uppermost sediment layers (Thomas et al. 1976). Another possibility

is that part of the NAP sedimented at this location is of natural origin. It should also be noted that the shapes of the sedimentation profiles are highly dependent on the correct identification of the 1935 horizon.

The sedimentation profiles of stations 3 and 6, especially the latter station, appear to lag behind the loading curve. It is unlikely that vertical migration of phosphorus is significant at these two stations. Rates of sedimentation are higher than at station 2 (Table 1), and, as Tzur (1971) for example has shown, the degree of modification of a sedimentation profile by vertical migration is inversely related to rate of sedimentation, other factors being equal.

A literal interpretation of the sedimentation profiles of stations 2, 3, and 6 suggests that, as anthropogenic phosphorus loadings increased, rate of NAP sedimentation increased initially only at station 2, later (around 1955) at station 3, and finally (around 1960) at station 6. The order 2, 3, 6 is the order of increasing distance from the major sources of anthropogenic phosphorus towards the western end of the lake. Thomas and Jaquet (1976) have shown that mercury sedimented in the Western basin is subsequently transported eastwards via the Pelee Passage into the Central basin. Most NAP entering Lake Erie is removed from the water column in the Western basin but accumulates in the sediments of the Eastern basin (Burns 1976; Burns et al. 1976). It is possible, therefore, that the NAP profiles of stations 3 and 6 reflect a gradual movement of phosphorus-rich sediment eastwards throughout the lake. The subject is discussed further by Burns et al. (1976).

The NAI-P values of the 0–1-cm horizons of cores 4 and 5 were more than twice as great as the values in the corresponding 1–2-cm horizons. The time spanned by the 0–1-cm horizons was not greater than 10 yr (Fig. 3). The thickness of the oxidized microzone at these two locations at the time of sampling was about 1 cm (J. D. H. Williams unpublished data). These facts suggest that the NAIP profiles of these two cores were appreciably influenced by migration of orthophosphate from the reduced zone into the oxidized microzone. This is possible as the rate of sedimentation at stations 4 and 5 was appreciably less than at the remainder (Table 1).

In a comparison study (Williams et al. 1976) 47 samples from the top 3 cm of the sediment column were analyzed. In many cases these samples straddled the boundary between oxidized microzone and the reduced zone below. At stations 1 to 6 this boundary was about 1 cm below the sediment–water interface (J. D. H. Williams unpublished data). In Williams et al. (1976) the question arose whether the statistical relationships between NAI-P and Fe_T or conc. HCl-Fe applied to oxidize and reduced zone alike. This study has shown that the difference between the NAI-P values of the 0–1, 1–2, and 2–4 cm layers of a given core, though often considerable, is minor compared with the widely ranging values of NAI-P in the 47 samples. This is why, in that study, possible differences in the NAI-P values of the oxidized and reduced zones were neglected.

Although some features of the sedimentation profiles are obscure this study has established that the changing anthropogenic phosphorus loadings of Lake Erie in recent years are reflected in the NAIP and organic phosphorus contents of contemporaneously deposited sediments. It can be assumed that the bulk of this phosphorus reaches the sediment–water interface as planktonic debris. Part of this phosphorus is apparently incorporated into the sediment column as organic phosphorus compounds. The remaining phosphorus in the debris is liberated as orthophosphate or soluble organic phosphorus compounds due to microbial decomposition of the debris at the interface. It is likely that the bulk of the NAIP of the oxidized microzone results from the reaction of the liberated orthophosphate with amorphous hydrated ferric oxide (Williams et al. 1976). It is likely also that some of the liberated orthophosphate is regenerated into the overlying water. If this is the case a cyclic process can be envisaged, in which part of the liberated orthophosphate is once again incorporated into living organisms and ultimately resedimented, so that at each pass progressively more of the phosphorus is removed from the water column. It is conceivable that the mean residence time of phosphorus in the water column can be related to the number of cycles a phosphorus atom undergoes on average.

The question has been raised whether, as a result of larger amounts of phosphorus in the sediments accumulating nowadays, the excess phosphorus could result in an increased return flow of phosphorus from sediments to overlying water under conditions favoring this process. The rate of regeneration of phosphorus is highly dependent on the oxygen status of overlying hypolimnetic waters. Burns and Ross (1972) estimated that regeneration was 11 times faster under anoxic than under oxic conditions in the Central basin of Lake Erie in the summer of 1970. Conversion of sedimented organic phosphorus into orthophosphate is probably not accelerated, and is very likely decreased, by a transition from oxic to anoxic conditions. Rate of dissolution of apatite is not affected by the Eh or oxygen status of the

solution it is immersed in, and, anyway, the mineral is essentially insoluble in Lake Erie (see above). It is the NAIP fraction, shown to be associated with iron (Williams et al. 1976) which accounts for the high rate of regeneration of phosphorus under anoxic conditions. The rate is probably dependent, among other factors, on the total amount of NAIP present in the uppermost sediment layers.

It is not so easy to come to any conclusions regarding regeneration rates under oxic conditions. The discussion above suggests that this rate may depend in large part on the relative amounts of amorphous hydrated ferric oxide and of orthophosphate at the sediment–water interface. Evidence from a wide variety of Wisconsin lakes, both oligotrophic and eutrophic, suggests that lake sediments are able to retain orthophosphate effectively up to a point where the atomic ratio of phosphate-reactive iron to NAIP is about 7 (Williams et al. 1971). A value of 31 was estimated for the same ratio for Lake Erie sediments (Williams et al. 1976). This apparent excess of phosphate-retaining sites may be the main reason why over 80% of input phosphorus is sedimented within the lake, even at present day loadings.

Obviously, the ratio of phosphate-reactive iron to NAIP is not the sole or even necessarily the most important factor controlling rate of regeneration of phosphate from sediments under oxic conditions; factors such as depth of overlying water and vigor of water circulation are almost certainly more important. However, the ratio is probably wide enough in Lake Erie to ensure that regeneration of orthophosphate from the sediments is not sufficient to markedly counteract any lowering of orthophosphate concentrations in the overlying water due to reduced anthropogenic phosphorus loading, provided the bottom waters are oxic at all times.

Summary and Conclusions

Sediment cores were taken from six localities throughout Lake Erie. Sedimentation rates at these localities varied from 0.8 to 7.6 mm per annum. Samples were also taken of the eroding bluffs along the northern shore of the lake, which contribute most of the sediment load to the lake. The materials were analyzed for particle-size distribution and for content of apatite phosphorus, nonapatite inorganic phosphorus (NAIP), and organic phosphorus. The estimates were denoted by the expressions apatite-P, NAI-P, and organic P. Estimates of nonapatite phosphorus (NAP) content were denoted by the expression NA-P and obtained by summing NAI-P and organic P values for a given sample. The analytical procedures used were modifications of procedures used to characterize modes of chemical combination of phosphorus in terrestrial soils.

At each of the six stations, conditions of sedimentation were essentially constant throughout the time periods of a few hundred years spanned by the cores examined here. The exception was one location (Station 4) where the most recently deposited sediment was more fine grained in texture than in the remainder of the core.

The total P values of the bluff samples averaged 576 $mg.kg^{-1}$, about 95% of this being apatite phosphorus. Although this source contributes a major portion of the total phosphorus entering Lake Erie the insolubility of apatite results in very little of this phosphorus being available for algal growth. This explains the difference in the water quality of the Eastern compared with the Western and Central basins, which receive large inputs of anthropogenic phosphorus in other chemical forms.

Lakewide, apatite phosphorus accounted for between 60 and 80% of total sedimented phosphorus prior to 1850. Apatite within Lake Erie sediments deposited within the last few hundred years is apparently exclusively of detrital origin: there is no interconversion between apatite and other forms of phosphorus within this time, as indicated by the constancy of apatite phosphorus concentrations with depth. The same constancy indicates a natural and not anthropogenic origin for this phosphorus.

On ascending the sediment cores, NAI-P and organic P values increased steadily with diminishing distance from the sediment–water interface in material deposited since 1850. At stations 4 and 5 this increase could be attributed to decreasing mean grain size of the sediment plus possible migration of phosphorus from the reduced zone into the oxidized microzone immediately below the sediment–water interface. At the remaining four stations, however, the increases were related to increasing rates of sedimentation of phosphorus of anthropogenic origin in recent years. At these four stations there was no eivdence that vertical migration of phosphorus up the profile was sufficient to significantly modify the nonapatite phosphorus (NAP) profiles of the cores; however, the cores from the more easterly locations suggested that increases in NAP sedimentation at these locations lagged by several years behind the recent increases in anthropogenic phosphorus loading. The reason for this is not clear, but may be related to difficulties encountered in drawing up separate phosphorus budgets for each basin of the lake (Burns et al. 1976).

Despite the increased phosphorus loadings of recent years, it was concluded that sufficient iron is co-sedimented with phosphorus in Lake Erie to render regeneration of orthophosphate from the sediments under oxic conditions an insignificant factor in the response of the lake to reduced phosphorus loadings.

Acknowledgments

We are indebted to A. L. W. Kemp for information and discussion on rates of sedimentation in Lake Erie and to several colleagues at C.C.I.W. for discussion of aspects of this study. J. T. Roe and F. H. Don, of the Diving Unit, C.C.I.W., T. Gantert and J. Shields collected the cores. Members of the Technical Operations Section, C.C.I.W., and the officers and crew of the C.S.S. *Limnos* participated in the sediment sampling program. We are grateful to J. W. Morse for his valuable criticisms of an early draft of this paper, and to P. G. Sly for permission to publish. We thank M. Stapleton for typing the manuscript.

BURNS, N. M. 1976. Nutrient budgets in Lake Erie, 1970. J. Fish. Res. Board Can. 33: 520–536.

BURNS, N. M., AND C. ROSS. 1972. Oxygen-nutrient relationships within the Central Basin of Lake Erie, p. 85–119. *In* N. M. Burns and C. Ross [ed.], Project Hypo. Can. Cent. Inland Waters Pap. 6.

BURNS, N. M., J. D. H. WILLIAMS, A. L. W. KEMP, J.-M. JAQUET, AND D. C. L. LAM. 1976. A phosphorus budget for Lake Erie. J. Fish. Res. Board Can. 33: 564–573.

DUNCAN, G. A., AND N. A. RUKAVINA. 1972. Abstracts, Conf. Great Lakes Res. 15: p. 82.

GARY, M., R. MCAFEE, JR., C. L. WOLF [ed.]. 1972. Glossary of geology. Washington, D.C. American Geological Institute. 805 p.

GELINAS, P. J., AND R. M. QUIGLEY. 1973. The influence of geology on erosion rates along the north shore of Lake Erie. Proc. Conf. Great Lakes Res. 16: 421–430.

GILBERTSON, M., H. H. DOBSON, AND T. R. LEE. 1972. Phosphorus and hypolimnial dissolved oxygen in Lake Erie, p. 141–145. *In* N. M. Burns and C. Ross [ed.] Project Hypo. Can. Cent. Inland Waters Pap. 6.

KEMP, A. L. W., R. L. THOMAS, C. I. DELL, AND J.-M. JAQUET. 1976. Cultural impact on the geochemistry of sediments in Lake Erie. J. Fish. Res. Board Can. 33: 440–462.

LEACH, J. H., AND S. NEPSZY. 1976. The fish community in Lake Erie. J. Fish. Res. Board Can. 33: 622–638.

PALACHE, C., H. BERMAN, AND C. FRONDEL. 1951. Dana's system of mineralogy, Vol. 2, 877–889. John Wiley & Sons, Inc., New York, N.Y.

ROBBINS, J. A., AND E. CALLENDER. 1975. Diagenesis of manganese in Lake Michigan sediments. Am. J. Sci. 275: 512–533.

SLY, P. G. 1976. Lake Erie and its basin. J. Fish. Res. Board Can. 33: 355–370.

THOMAS, R. L., A. L. W. KEMP, AND C. F. M. LEWIS. 1972. Distribution, composition and characteristics of the surficial sediments of Lake Ontario. J. Sediment. Petrol. 42: 66–84.

THOMAS, R. L., AND J.-M. JAQUET. 1976. Mercury in the surficial sediments of Lake Erie. J. Fish. Res. Board Can. 33: 404–412.

THOMAS, R. L., J.-M. JAQUET, A. L. W. KEMP, AND C. F. M. LEWIS. 1976. Surficial sediments of Lake Erie. J. Fish. Res. Board Can. 33: 385–403.

TZUR, Y. 1971. Interstitial diffusion and advection of solute in accumulating sediments. J. Geophys. Res. 76: 4208–4211.

WILLIAMS, J. D. H., AND T. MAYER. 1972. Effects of sediment diagenesis and regeneration of phosphorus with special reference to lakes Erie and Ontario, p. 281–315. *In* H. E. Allen and J. R. Kramer [ed.]. Nutrients in natural waters. Wiley-Interscience, New York, N.Y.

WILLIAMS, J. D. H., J.-M. JAQUET, AND R. L. THOMAS. 1976. Forms of phosphorus in the surficial sediments of Lake Erie. J. Fish. Res. Board Can. 33: 413–429.

WILLIAMS, J. D. H., J. K. SYERS, S. S. SHUKLA, R. F. HARRIS, AND D. E. ARMSTRONG. 1971. Levels of inorganic and total phosphorus in lake sediments as related to other sediment parameters. Environ. Sci. Technol. 5: 1113–1120.

WILLIAMS, J. D. H., J. K. SYERS, T. W. WALKER, AND R. SHAH. 1969. Apatite transformations during soil development. Agrochimica 13: 491–501.

WILLIAMS, J. D. H., AND T. W. WALKER. 1969. Fractionation of phosphate in a maturity sequence of New Zealand basaltic soil profiles: 2. Soil Sci. 107: 213–219.

Cultural Impact on the Geochemistry of Sediments in Lake Erie

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The chemical and physical characteristics of six cores from Lake Erie are described. The concentrations of Si, Al, K, Na, and Mg, which represent the major mineral species in the sediment, were uniform in each core. Surface enrichments of Hg, Pb, Zn, Cd, Cu, Organic-C, N, and P were observed at each location, due mainly to the increasing anthropogenic loading of these elements to the sediments since 1850. Concentration profiles for Mn, Fe, and S were related to sediment Eh and are due to mobilization of these elements in the pore waters.

Natural and anthropogenic inputs of nutrients and heavy metals were calculated. About 60% of the total loading of these elements is being deposited in the Eastern basin. It is calculated that 30 million metric tons of fine-grained sediment accumulates in the offshore basins of the lake, each year. The major source of the sediment input is erosion of the shoreline bluffs, with the north shore between Erieau and Long Point contributing 21 million metric tons annually. A sediment mass balance suggests that a large part of this bluff material is transported up to 150 km and is deposited in the Eastern basin.

KEMP, A. L. W., R. L. THOMAS, C. I. DELL, AND J. -M. JAQUET. 1976. Cultural impact on the geochemistry of sediments in lake Erie. J. Fish. Res. Board Can. 33: 440–462.

Les auteurs décrivent les caractéristiques chimiques et physiques de six carottes prélevées dans le lac Erié. Les teneurs en Si, Al, K, Na et Mg, représentant les principaux minéraux du sédiment, sont uniformes dans chaque carotte. L'enrichissement de la surface en Hg, Pb, Zn, Cd, Cu, C organique, N et P a été observé à chaque endroit, causé en grande partie par un apport anthropogénique croissant de ces éléments dans les sédiments depuis 1850. Les profils de teneur en Mn, Fe et S sont liés à l'Eh du sédiment et sont le résultat de la mobilisation de ces éléments dans l'eau des pores.

Les apports naturels et anthropogéniques de substances nutritives et de métaux lourds ont été calculés. Environ 60% de la charge totale est déposé dans le bassin de l'est. On calcule que 30 millions de tonnes métriques de fin sédiment s'accumulent annuellement dans les bassins du large. La principale source de l'apport de sédiment est l'érosion des falaises qui bordent le lac, la rive nord entre Erieau et Pointe Longue contribuant 21 millions de tonnes métriques annuellement. Un bilan de masse du sédiment donne à croire qu'une grande partie de ce matériel de falaise est transporté jusqu'à 150 km de distance et est déposé dans le bassin de l'est.

Accepted November 17, 1975

Accepté le 17 novembre 1975

CHANGES in the environment and biota of Lake Erie over the last 100 yr have been well documented (eg. Beeton 1965, 1969; I.J.C. 1969). The accelerated eutrophication of the lake is attributed to human activities since the time of settlement of the drainage basin. Evidence of change in the last 100 yrs has been dependent mainly on water intake data and commercial fish catches, together with only a few scientific studies (Beeton 1969). There is only meager information available on the increasing inputs of materials to the lake since the settlement of the basin. Clues as to the changing limnological conditions of the lake are, however, preserved in the sediments.

Printed in Canada (J3997)
Imprimé au Canada (J3997)

Lake Erie is ideally suited for a study of its recent limnological history from sediment cores, as rates of sedimentation are extremely high (Kemp et al. 1974), allowing preservation of the historical record. Recent studies have demonstrated the changing environment of the lake. Enrichment of organic carbon (Org.-C), nitrogen (N), and phosphorus (P) in the surface sediments since around 1900 is attributed mainly to increasing nutrient inputs to the lake (Kemp 1971; Williams and Mayer 1972; Kemp et al. 1972, 1974; Williams et al. 1976a). Increases in mercury (Hg) concentration in the Erie surface sediments are also due to recent Hg loading increases to the lake (Walters et al. 1972; Kovacik and Walters 1973; Walters et al. 1974; Wolery and Walters 1974; Kemp et al. 1974). Two major

increases in Hg concentration in the surface sediments from the Western basin of Lake Erie are related to the building of chlor-alkali plants in 1939 and 1954 (Wolery and Walters 1974). Significant enrichments of chromium (Cr), cadmium (Cd), zinc (Zn), nickel (Ni), antimony (Sb), arsenic (As), copper (Cu), and Hg are observed in the top 30 cm from eight cores from Lake Erie (Walters et al. 1974). The enrichments are greatest in cores from Buffalo and Cleveland harbors.

The sediments act as a major sink for incoming materials to Lake Erie. For example, it is estimated that 84% of the total P entering Lake Erie and 56% of the total N is retained in the lake (I.J.C. 1969). Most of the P and N is retained in the lake sediments. In order to properly understand the elemental geochemical cycles in Lake Erie, it is necessary to study the sources, dispersion pathways, and sinks of the sedimentary material. Only in the cases of the Hg-, S-, and P-cycles in Lake Erie, is such an understanding coming close to realization.

The sources and sinks of Hg have been discussed by Kovacik and Walters (1973), Wolery and Walters (1974), and Kemp et al. (1974). The dispersion pathways of Hg were discussed by Thomas (1974), Walters and Herdendorf (1975), and by Thomas and Jacquet (1976). In a comprehensive study of the S-cycle in Lake Erie, Nriagu (1975) demonstrated that 60–70% of the sulphate concentration in the lake is of anthropogenic origin. Sulphate inputs to the lake have increased steadily since 1850 and have only levelled off in the last decade. A mass balance for S in Lake Erie showed that only 2% of the S input to the lake is retained in the sediments. The Lake Erie P-cycle is discussed by Williams et al. (1976a, b) and Burns (1976a, b), in this issue and will not be discussed further.

This report outlines the vertical distribution of organic matter, major elements, trace elements, and minerals at six offshore locations in Lake Erie. The natural background (pre-settlement) levels of geochemical elements and mineral species are compared with those found in 12 shoreline bluff locations and with the post-settlement (i.e. post 1850) sediment levels. An attempt is made to explain the surficial geochemistry in terms of chemical and physical processes, based on the six cores. The potential sources and dispersion pathways of the Lake Erie sediments are also discussed. On the basis of our results, preliminary estimates are made on the natural and anthropogenic loadings of nutrients and trace metals to the Erie offshore sediments.

Methods

Sediment cores were collected at 10 sample locations (Fig. 1 and Table 1). Site selection was determined on the basis of lakewide echosounding profiles, the six main sample locations being close to the zone of maximum postglacial sediment thickness in each basin. Sampling at these locations was carried out in August 1970 (G-16), May 1971 (U-42, W-44, T-44, B-8), and July 1971 (M-32). *Ambrosia* and *Castanea* pollen counts (Anderson 1974) were determined on these cores together with four supplementary cores (E-11, L-19, H-21, and N-34) collected in 1971 for the P study described in this issue (Williams et al. 1976a). Shoreline bluff samples were collected at 12 sites on the Ontario shore of the lake (Fig. 1), where rapid shoreline erosion is presently taking place (Kemp and Dell 1975). Fine-grained glacial tills or clays were collected from each bluff site.

The sediment cores were obtained with a modified

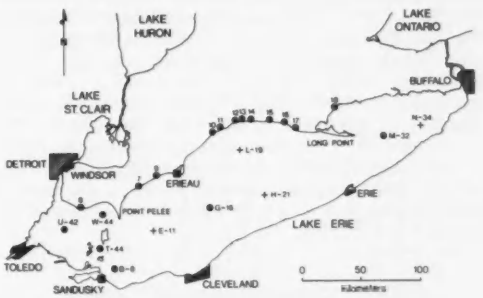

FIG. 1. Sediment and bluff sample locations in Lake Erie.

TABLE 1. Present-day sedimentation rates at the 10 sample locations in Lake Erie.

Station number	Water Depth (*m*)	Present-day sedimentation rate ($g\ m^{-2}\ yr^{-1}$)	Mean annual accumulation of sediment (*mm*)
		Western Basin	
U-42	10	3580	7.6
W-44	10	3456	6.7
T-44	11	845	1.1
		Central Basin	
B-8	14	1110	2.7
E-11	21	525	2.8
G-16	25	1190	5.1
L-19	22	395	0.8
H-21	23	555	0.8
		Eastern Basin	
M-32	58	5050	13.4
N-34	29	2155	3.3

benthos gravity triple corer (Kemp et al. 1971). Eh and pH values were measured within 2 h of retrieval of the triple corer, on one of the cores. Sediment from the same core and a second core was then immediately freeze dried and stored for geochemical analysis. The remaining core was stored at 4 C and subsampled later for analysis of sediment particle size and *Ambrosia* and *Castanea* pollen content. Subsampling of the cores, sample preparation, Eh and pH measurements, determination of water content, pollen counts, organic carbon, carbonate carbon, total nitrogen, mercury and sediment particle size have been previously described (Kemp et al. 1972, 1974).

Major element analysis for total Na, K, Ca, Mg, Fe, Mn, P, S, Ti, Al, and Si was carried out by X-ray fluorescence, using a Philips PW 1220C semi-automatic X-ray fluorescence spectrometer on pelletized samples. Trace elements, extractable in concentrated HCl, were determined on a Varian AA-5 atomic absorption spectrophotometer. The freeze dried sediment samples were extracted in hot concentrated HCl (~ 90 C) for 20 min and the leachate analyzed for Pb, Cu, Zn, Cd, Be, and V. The extraction method was tested against other extraction methods (e.g. aqua regia, conc. nitric acid) and found to yield reproduceable results. The method extracts the same quantity of the trace metals from the sediment as aqua regia (B. E. St. John personal communication).

The mineralogical composition of surface sediment samples (0–1 cm) and selected subsurface sediment samples was determined by microscopic examination and X-ray diffraction.

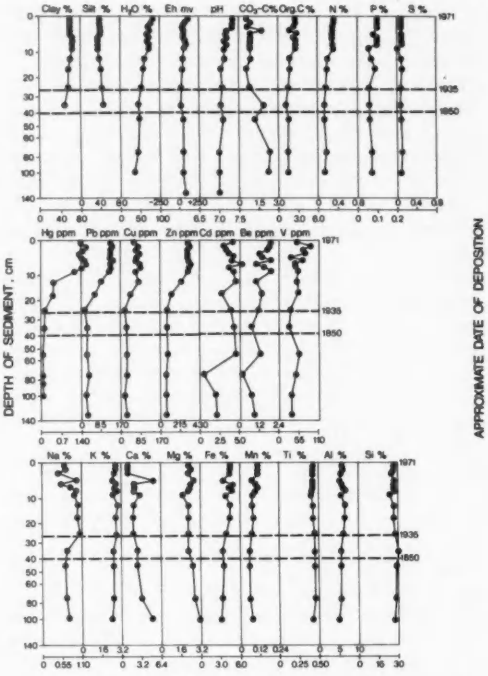

FIG. 2. Concentration vs. depth profiles for sediment constituents at station U-42, Lake Erie — Western basin.

Physical Characteristics and General Description of the Sediments

The fine-grained sediments in each of the three basins are generally similar in their physical characteristics, with the exception of the redox potential (Eh) which varies depending on the hypolimnetic oxygen concentration. This is mainly due to the more or less uniform source of inorganic sediment materials from the drainage basins. The sediments at each location consist characteristically of a reddish-brown floc from zero to a few millimeters thickness at the sediment–water interface overlying a soft gray to dark gray ooze. The sediment, which is soft in the top few centimeters, becomes increasingly firmer with depth of burial. The color and firmness of the sediments are reflected by water content and redox potential (Fig. 2–7). The water contents decreased from surface values of around 90% water by weight in the sediment to about 50–60% water at a depth of 1 m. Additionally, variations in water content in each core match the sediment particle size distribution, with the coarser sediments having lower water contents. The decrease in water content from the interface reflects the increasing compaction of the sediment with depth of burial.

The Eh is positive in the top cm of sediment at all locations, except at Station G-16, and decreases with depth of burial to around +0.025 V at 1 m (Fig. 2–7). Insertion of the electrode in the oxidized top mm of sediment yielded Eh values around +0.300 V, close to that of the bottom water. The sediment color and Eh indicate that the uppermost few mm of positive Eh contain hydrated iron oxides, whereas the black reduced zones contain iron sulphides (Hough 1958). Exposure of the dark gray reduced sediment to air oxidizes the dark gray sediment to a brown color. The hypolimnion waters of the central basin of Lake Erie were anoxic at the time of the collection of the core at Station G-16 in August 1970. It can be seen that the Eh is negative at the sediment–water interface and is most reduced between 1 and 3 cm (Fig. 6). A core collected at the same location in May 1971, when the hypolimnion waters were saturated with oxygen, yielded an Eh profile similar to that of Station M-32 (Fig. 7). This demonstrates the seasonal

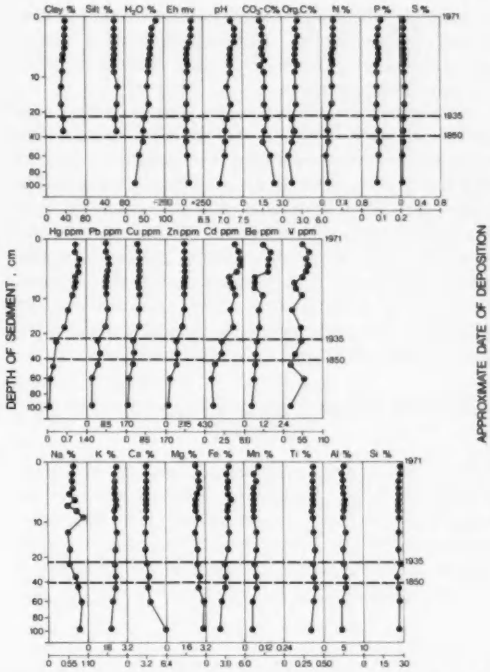

FIG. 3. Concentration vs. depth profiles for sediment constituents at station W-44, Lake Erie — Western basin.

variations possible in sediment Eh near the water interface. Oxygen depletion of the hypolimnion water in the Central and Western basins (Beeton 1965) will affect the chemical composition of the surface sediments in these basins. Mobilization of Mn and Fe is particularly important under reducing conditions (Mackereth 1966) and is discussed in the next section. More extensive discussion of redox potential in the Great Lakes surface sediments may be found in Kemp and Lewis (1968), Mortimer (1971), Vanderpost (1972), and Kemp et al. (1972).

Sediment pH decreases from values of around 7.3 at the sediment–water interface to uniform values around 7.0 at 1 m in the cores from Station U-42, W-44, T-44, and M-32 (Fig. 2, 3, 4, and 7). The pH is uniform around 7.0 at Station G-16 (Fig. 6) and increases from a surface value of 6.8 to a value of 7.1 at Station B-8 (Fig. 5). The decrease of pH values in the cores is believed to be due to decomposition of sediment organic matter and release of CO_2 and NH_4^+ to the interstitial waters (Kemp et al. 1972). It is not clear why the pH values are uniform or increase with depth in the two Central basin cores.

The sediment particle size distribution is uniform at all locations except Stations T-44 and B-8 (Fig. 2–7). The sediments consist of 50–75% clay-size materials (<0.004 mm) with silt-size sediment (0.004–0.062 mm) comprising most of the remainder. The sediment particle size is dependent on the water depth at these offshore locations, with the finest sediments being found in the deeper eastern locations. The abrupt change in sediment particle size at Stations T-44 and B-8 (Fig. 4 and 5) may be related to some event in the lake's recent history; for example, lower energy conditions due to higher water levels and/or a large storm in the lake. The nonuniformity of the sediment particle size distribution at these two locations and the chemical profiles at other locations is indicative of sedimentation conditions in the lake varying with time.

The surface sediments consist mainly of quartz, feldspars, organic matter, and clay minerals.

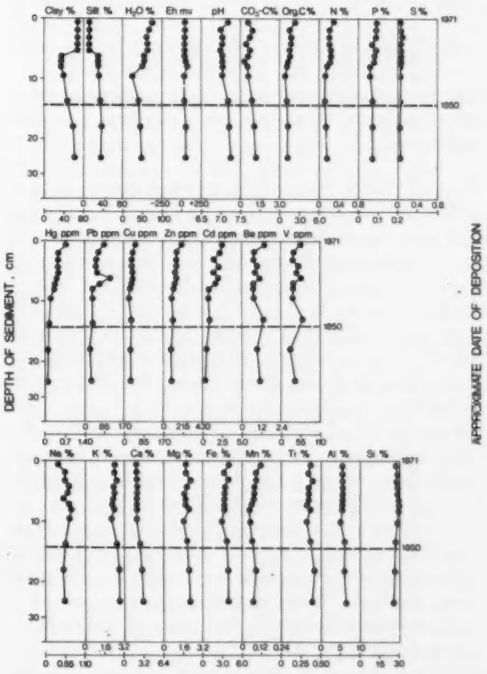

FIG. 4. Concentration vs. depth profiles for sediment constituents at station T-44, Lake Erie — Western basin.

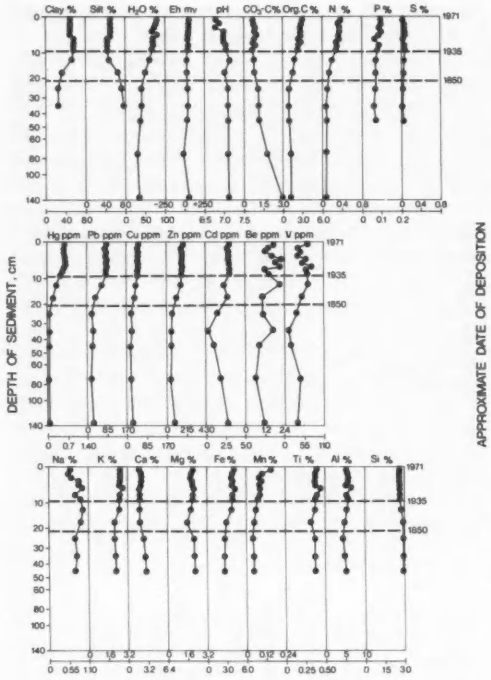
Clay % Silt % H2O % Eh mv pH CO2-C% Org.C % N % P % S %
Hg ppm Pb ppm Cu ppm Zn ppm Cd ppm Be ppm V ppm
Na % K % Ca % Mg % Fe % Mn % Ti % Al % Si %
1971
1935
1850
DEPTH OF SEDIMENT, cm
APPROXIMATE DATE OF DEPOSITION

FIG. 5. Concentration vs. depth profiles for sediment constituents at station B-8, Lake Erie — Central basin.

Thomas (1969) demonstrated a high degree of correlation between the clay size fraction and clay mineral contents of fine-grained Great Lake sediments and it is believed that a similar situation exists at these locations. The clay minerals consist of around 70–80% illite [$KAl_2(AlSi_3O_{10})(OH)_2$] and 20–30% chlorite [$Mg_5Al(AlSi_3O_{10})(OH)_8$] and kaolinite [$Al_4Si_4)_{10}(OH)_8$]. Dolomite [$CaMg(CO_3)_2$] and calcite [$CaCO_3$] abundances are low, calcite being less than dolomite. Vivianite, an iron phosphate, is also present in the surface sediments. The occurrence and origin of this diagenetic mineral in the lake has been described by Dell (1973a) and Nriagu and Dell (1974).

In addition to the naturally occurring minerals in the sediments, there is also an appreciable amount of anthropogenic materials in the sand sized fraction. These grains occur in both the light and heavy fractions, but more abundantly in the heavy portion. An examination of these grains show that they take on a variety of forms (Table 2). The greatest numbers occur at Stations B-8 and G-16 in the central basin. Station M-32, although it has the smallest numbers of the black or brown types, contains a great many grains of a peculiar white, somewhat porous material. Preliminary examination suggests that it is not a naturally occurring material and that it is probably anthropogenic.

Qualitative examination of these grains suggests that each location has its own assemblage of anthropogenic materials. For example, Station B-8 has a noticeable number of black, submetallic irregular grains with a conchoidal fracture, and Station G-16 is characterized by black, metallic, somewhat porous grains. These preliminary observations are being further pursued in our laboratories. It is possible that these particles can be related to the anthropogenic chemical loading to the lake and also possibly to their sources.

Chemical Composition and Enrichment of the Sediment

The chemical composition of the cores is shown in Fig. 2–7. A number of patterns emerge on examination of the chemical profiles. The concentration of a number of the elements is much greater near the sediment–water interface than at the *Ambrosia* horizon (~ 120 yr BP). Hg, Pb, Zn,

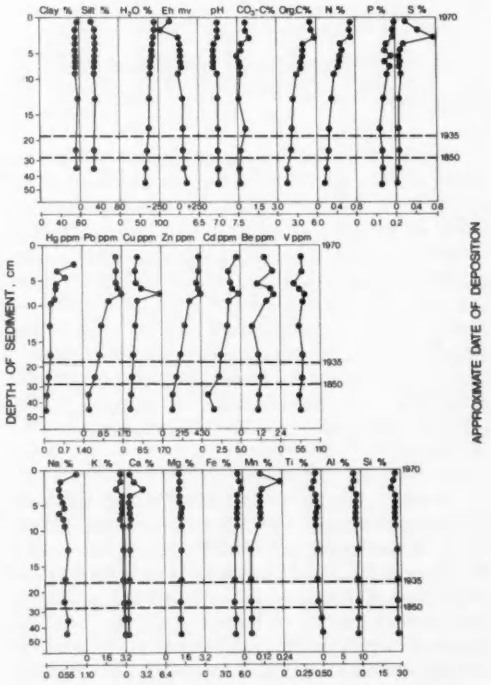
Clay % Silt % H2O % Eh mv pH CO2-C% Org.C % N % P % S %
Hg ppm Pb ppm Cu ppm Zn ppm Cd ppm Be ppm V ppm
Na % K % Ca % Mg % Fe % Mn % Ti % Al % Si %
1970
1935
1850
DEPTH OF SEDIMENT, cm
APPROXIMATE DATE OF DEPOSITION

FIG. 6. Concentration vs. depth profiles for sediment constituents at station G-16, Lake Erie — Central basin.

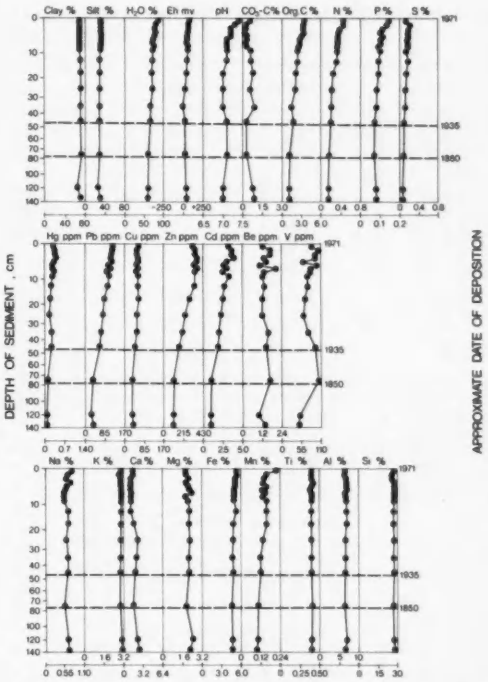
Clay % Silt % H2O % Eh mv pH CO3-C% Org.C % N % P % S %
Hg ppm Pb ppm Cu ppm Zn ppm Cd ppm Be ppm V ppm
Na % K % Ca % Mg % Fe % Mn % Ti % Al % Si %
1971
1935
1850
DEPTH OF SEDIMENT, cm
APPROXIMATE DATE OF DEPOSITION

FIG. 7. Concentration vs. depth profiles for sediment constituents at station M-32, Lake Erie — Eastern basin.

Cd, Cu, Org.-C, N, P, Mn, and Fe all show this characteristic to varying degrees. On the other hand, Si, Al, K, and Ti are generally uniform in the cores. A slight decrease in the concentration of these elements however, is observed near the sediment–water interface, compensating for the variations in organic matter and carbonates near the interface.

SEDIMENT ENRICHMENT FACTOR

It has already been shown that the sediment consists mainly of clay minerals, quartz, carbonates, and feldspars and it can be seen that Si, Al, Fe, K, Na, Ca, and Mg are the dominant inorganic elements in the sediment matrix. Clay minerals form the largest part of the sediment material at these fine-grained locations and account for most of the aluminum in the sediments. Therefore, to normalize the data, the concentrations of each element should have the same ratio to aluminum as that found below the *Ambrosia* horizon.

The normalized data are shown for each element in Table 3 and 4. To quantify the results, we have introduced a ratio that is designated as the Sediment Enrichment Factor (SEF), which is defined below:

$$SEF = \frac{\frac{E_s}{Al_s} - \frac{E_A}{Al_A}}{\frac{E_A}{Al_A}}$$

where E_s = The observed elemental concentration in the surface cm of sediment
E_A = The observed elemental concentration below the *Ambrosia* horizon
Al_s = The aluminum concentration in the surface cm of sediment
Al_A = The aluminum concentration below the *Ambrosia* horizon

The SEF is then a measure of the degree of normalcy of the most recent sediment being deposited. Zero values represent elements with constant concentration in the sediment. Positive values indicate increased concentration in the surface sediment, the size of the SEF representing the degree of concentration increase. Negative values indicate loss of the element from the sediment matrix. For comparative purposes, the elements are arranged into six separate groupings. each having similar SEF trends (Table 3 and 4).

The elements that make up the bulk of the sediment matrix and are unlikely to be affected by diagenesis or increasing eutrophication to the lake are defined as the Conservative group of elements (Si, K, Ti, Na, and Mg). Aluminum which was used to normalize the data is also part of this group. The Enriched elements (Hg, Pb, Zn, Cd, and Cu) and the Nutrient elements (Org.-C, N, and P) are all characterized by higher concentrations at the sediment–water interface relative to the *Ambrosia* horizon. The Carbonate group of elements (CO_3-C and Ca) have a similar SEF pattern at each location showing enrichment or loss, and are grouped together as one unit. The Mobile elements, (Fe, Mn, and S) like the Enriched ones, have higher surface concentrations, but have concentration profiles that are related to the physical conditions of the sediment. The Miscellaneous elements (Be and V) have no characteristic concentration profiles and are grouped under one heading for convenience.

The core data, consisting of 31 variables measured on 74 samples, were split into the following four groups: All samples; Recent samples; early-colonial samples; and precolonial samples for all the lakes.

Mean, standard deviation, and correlation coefficient matrices were computed for each group, followed by R-mode factor analysis (Cameron

TABLE 2. Abundance and description of the anthropogenic materials in Lake Erie surface sediments.

Station number	Abundance of anthropogenic materials	Color	Luster	Shape	Structure
U-42	Moderate abundance	Black	Submetallic	Irregular	Conchoidal fracture, somewhat glassy looking
		Black	Submetallic	Irregular	Vesicular
		Buff	Dull	Very irregular	Vesicular
W-44	Moderate abundance	Black	Submetallic	Irregular	Vesicular
		Shades of brown or white	Dull	Very irregular	Globular aggregates
B-8	Abundant	Black	Metallic	Irregular	Massive
		Dark grey	Metallic	Irregular	Slightly fibrous looking
		Black	Submetallic	Irregular	Vesicular
		Black	Submetallic	Irregular	Conchoidal fracture, somewhat glassy
		Shades of buff or white	Dull	Very Irregular	Vesicular
		Shades of buff, brown, or rust color	Dull	Irregular	Globular aggregates
G-16	Abundant	Black	Metallic	Irregular	Vesicular
		Brown	Dull or shiny	Irregular	Globular aggregates
M-32	White components abundant, dark components minor	White	Dull	Irregular	Semiporous
		Black	Submetallic	Irregular	Vesicular

TABLE 3. Sediment enrichment factors (SEF) for the Conservative, Enriched, and Nutrient elements. Values between −0.2 and +0.2 are designated as zero.

Station number	Conservative elements					Enriched elements					Nutrient elements		
	Si	K	Ti	Na	Mg	Hg	Pb	Zn	Cd	Cu	Org.-C	N	P
						Western Basin							
U-42	0	0	0	0	0	22.8	3.2	2.6	1.4	1.4	0.8	1.3	0.4
W-44	0	0	0	0	0	4.3	0.9	0.7	1.9	0.6	0.5	1.0	0
T-44	0	0	0	0	0	13.2	3.4	1.7	4.0	1.4	1.1	1.2	0.5
						Central Basin							
B-8	0	0	0	−0.3	0	9.7	2.6	1.4	0.8	1.1	1.3	2.1	0.3
G-16	0	0	0	0	0	11.4	6.7	3.2	4.6	1.2	2.8	3.0	0.7
						Eastern Basin							
M-32	0	0	0	0	0	7.3	3.7	2.2	2.7	0.8	1.7	2.0	0.8

TABLE 4. Sediment enrichment factors values for the Carbonate, Mobile, and Miscellaneous elements.

Station number	Carbonate elements		Mobile elements			Miscellaneous elements	
	(CO_3-C	Ca)	(Fe	Mn	S)	(Be	V)
			Western Basin				
U-42	−0.5	0	0	0.4	0	0.4	0
W-44	0	0.4	0	0	0	0	0
T-44	−0.4	−0.3	0	0.4	0.3	1.0	1.5
			Central Basin				
B-8	−0.4	−0.4	0.3	0.4	0	0.5	0.7
G-16	58.0	4.3	0.3	2.7	8.2	0.6	0.3
			Eastern Basin				
M-32	0	−0.3	0	1.1	1.0	0.3	0.7

1967). Interpretation of the results led to no general pattern, unlike the surface samples in Lake Huron (Thomas et al. 1973). This could be due to the small number of samples in the various groups, to near-invariance of same parameters with depth of burial (Fig. 2–7) and to depth-dependent diagenetic processes in the cores. Pending further samples, only the most obvious correlations will be discussed.

CONSERVATIVE ELEMENTS

As defined previously, Si, Al, K, Na, and Mg have been assigned to this grouping of elements. Their zero SEF values indicate that the concentration of these elements in the sediments have remained essentially constant. It can be seen from the profiles of these elements that their concentration fluctuates around a general uniform value in the sediment column (Fig. 2–7). The fluctuations are particularly striking for Na. Variations in the concentration of these elements in the cores, as with sediment particle size discussed previously, are again suggestive of nonuniform deposition of these major materials. With the exception of carbon, iron, and calcium, these elements are the most abundant in the sediment materials, reflecting the major mineralogical species derived from terrigenous sources.

A linear correlation matrix, based on all samples, for the Conservative elements and including silt- and clay-size fractions, which make up the bulk of the sediment, is given in Table 5. It can be seen that Al shows strong positive relationships to K and clay and a lesser positive relationship to Ti. This is in agreement with the earlier observation that Al is indicative of the clay mineral content, while the relationship of Al to K confirms the dominance of illite as determined by X-ray diffraction. The relationship of Al to Ti suggests that titanium is clay-lattice-bound, probably in illite as discussed by Dolcater et al. (1970).

Both Si and Na are poorly related to other

TABLE 5. Correlation coefficient matrix for Conservative elements (74 samples).

	Al	K	Ti	Na	Mg	Silt-size fraction	Clay-size fraction
Si	−.487**	−.615**	−.479**	.217	−0.91	.346*	−.641**
Al		.864**	.626**	−.361**	−.667**	−.529**	.754**
K			.638**	−.244*	−.367**	−.565**	.837**
Ti				−.021	−.131	−.178	.512**
Na					.360**	.360**	−.297**
Mg						.541**	−.470**
Silt-size fraction							−.840**

**Significant at 0.01 probability level.
*Significant at 0.05 probability level.

Conservative elements and no positive interpretation can be made with respect to them. Mg shows a good positive relationship with the silt-size fraction and a negative relationship with the clay-size fraction (Table 5). On the basis of the SEF values (Table 3) it is obvious that Mg is behaving as a Conservative element and is hence included in this grouping. However, the relationship of Mg to silt suggests that Mg representes the observed detrital dolomite contribution, and it is discussed subsequently under the section on Carbonate elements.

ENRICHED ELEMENTS

These elements form the smallest group in the sediment, accounting for less than 0.1% of the sediment materials. However, they are a most significant group as most of the metals are toxic in high concentrations and it is important to understand their geochemical cycles in the sediments (Wood 1974). The Enriched elements are characterized by small concentrations below the *Ambrosia* horizon and much higher concentrations near the sediment–water interface (Fig. 2–7).

Before discussing the results, it is necessary to examine the possible processes that can yield trace element profiles like those observed. Enrichment of a chemical element at the interface may be due to two possible factors: (1) Increased anthropogenic loading of the sediments in recent years, and (2) diagenesis and/or upward migration of chemical species in the pore waters (Kemp et al. 1972). As with the Hg profiles in Lake Erie (Kemp et al. 1974), the P profiles (Williams et al. 1976a), and the trace element profiles in Lake Erie (Walters et al. 1974), the enrichments are believed to be mainly due to the first factor. Similar conclusions have been drawn recently for the increases in Hg, Pb, Zn, Cu, and Cr in Lake Michigan (Leland et al. 1973); Pb, Zn, Cu, Cd, Ni, and Cr in the Wisconsin lakes (Iskandar and Keeney 1974); Cd, Pb, Zn, and Cu in the Baltic Sea (Erlenkeuser et al. 1974); and for Pb, Cr, Cd, Zn, Cu, Ag, V, and Mo in the southern California coastal sediments (Bruland et al. 1974).

The increase in elemental concentration appears to begin at around the level of the *Ambrosia* horizon at most of the locations, lending primary support to this argument. For example, it can be seen that the Pb and Hg concentration commences to increase from low baseline values at 17.5 cm at Station B-8 (Fig. 5), 25 cm at Station G-16, (Fig. 6), and 78 cm at Station M-32 (Fig. 7), coinciding with the *Ambrosia* horizon. The other Enriched elements generally follow the same trends at these locations. Heavy metals increase also at the *Ambrosia* horizon at Station T-44 (Fig. 4). Enrichment appears to commence above the horizon at Station U-42 (Fig. 2) and below at Station W-44 (Fig. 3).

Diagenetic-mobilization mechanisms have been proposed by a number of authors to explain exponential elemental concentration curves such as are observed for some of the elements in these cores (e.g. Manheim 1970; Berner 1971). These mechanisms assume steady-state conditions, which cannot be applied at most of our locations. The variable sedimentation rates, mentioned previously, the fluctuating concentration profiles and the exponential increase in anthropogenic loading to the lakes (Beeton 1969) preclude steady-state conditions. However, these factors do not rule out diagenetic mobilization mechanisms as a contributor to the elemental profiles observed in Lake Erie.

A comparison of the recent sediment, precolonial sediment and the bluffs (Table 6) also lends further support to the anthropogenic origin of the heavy metals. The results show that Pb, Cu, and Cd concentrations in the precolonial sediment match those of the bluffs which are the primary source materials. The high concentra-

TABLE 6. Average concentrations of the Enriched elements in the bluffs and the recent and pre-colonial sediments (Concentrations in μg/g dry weight of sediment).

Element	Recent sediments	Pre-colonial sediments	Bluff samples
No. of samples	6	6	17
Hg	0.855	0.075	0.045
Pb	106	28	35
Zn	279	98	52
Cd	4	1	1
Cu	57	29	27

tions of these metals in the recent sediments are clearly of anthropogenic origin. Hg and Zn appear to be enriched in the precolonial sediments as opposed to the bluffs. If mobilization and upward migration of these elements had occurred in the sediments, their concentrations in the precolonial sediments should have been depleted relative to the bluffs. The higher concentrations of Hg and Zn in the precolonial sediment are presumably a result of their concentration during weathering, transportation and deposition on the lake bed.

It is not possible to make any clear interpretation of the Enriched element profiles, because of fluctuations in the elemental concentration (Fig. 2–7). The concentration of these elements relative to the *Castanea* horizon would indicate that the major enrichment has taken place since 1935. Presumably the increases have taken place most extensively since 1950, paralleling the increases of Pb in the Greenland icecap (Murozumi et al. 1969).

The fluctuations in trace element concentration may represent variable sediment inputs or may in part be due to analytical precision. High sedimentation rates and small elemental concentrations can produce an irregular profile due to nonuniform deposition of the anthropogenic and natural materials. Wolery and Walters (1974) used changes in the Hg and Cr profiles to date increases in elemental loading to western Lake Erie. Trace element profiles in Buffalo and Cleveland harbor sediments were also erratic, indicating variable inputs of the metals (Walters et al. 1974). The authors noted also that diagenesis and mobilization of the metals might account partially for the irregularity of the profiles, as is probably the case in our cores.

The results of this study and of the previous studies by Walters et al. (1974) and Wolery and Walters (1974) suggest that the heavy metal enrichments in Lake Erie sediment cores are mainly of anthropogenic origin. The best evidence for anthropogenic inputs from this study is provided by the increases in heavy metal concentrations coinciding generally with the Ambrosia horizon, the comparison with the bluff samples and the observation of anthropogenic particles in the surface sediments. The large heavy metal inputs in Buffalo and Cleveland harbors (Walters et al. 1974), the stepwise Hg and Cr concentration increases in the Western basin (Wolery and Walters 1974) and the absence of Hg and nutrient enrichments in the Lake Huron cores (Kemp et al. 1974) all point towards an anthropogenic origin of the heavy metals. The fluctuating concentration profiles, which are probably a function of irregular anthropogenic inputs, are possibly also modified to a minor extent by diagenesis and migration of the metals. For this study, it is therefore assumed that the heavy metal enrichments are anthropogenic. Further research will be required to determine the extent of sediment enrichment of the heavy metals by diagenesis and mobilization.

Statistically, a high degree of correlation can be demonstrated between individual elements in the Enriched group and Org.-C (Table 7). The relationship of Pb, Zn, Cd, and Cu to Org.-C may be indicative of complexing and/or absorption by organic matter as suggested from southern Lake Michigan by Shimp et al. (1971) or may be a fortuitous relationship reflecting increased anthropogenic loading with time paralleled by increasing nutrient supply and associated primary productivity in the lakes.

The distribution of the heavy metals in the sediment column may also have been modified by mixing of the uppermost layers of sediment. A recent study of ^{137}Cs and ^{210}Pb in cores from Lake Michigan demonstrated that mixing occurred to depths of 1–4 cm (Robbins and Edgington 1975). These cores came from offshore locations at water depths greater than 80 m. The mixing depths are similar to those estimated for offshore locations in lakes Ontario, Erie, and Huron (Kemp et al. 1972). In this study, where the majority of the sample locations are in much shallower water, mixing could be a factor in modifying surface enrichment and the concentration profiles of the chemical elements.

Mixing may be due to: (1) bioturbation near the interface, (2) gas movement from the underlying sediment, and (3) turbulent mixing (Lee 1970). Bioturbation and gas movement may be expected in the western basin of Lake Erie. Tubificid populations close to the mouth of the Detroit River near Station U-42 have higher values than for any other region of the Great Lakes (Brinkhurst et al. 1968). Methane gas release has also been demonstrated for Lake Erie sediments, with the highest release in the western

TABLE 7. Correlation coefficient matrix for Enriched elements (74 samples).

	Hg	Pb	Zn	Cd	Cu
Org.-C	.245*	.818**	.843**	.715**	.654**
Hg		.398**	.216	.590**	.334**
Pb			.903**	.773**	.805**
Zn				.712**	.777**
Cd					.756**

basin (Mallard and Frea 1972). Examination of the chemical profiles suggests that mixing may be taking place at the shallower of the offshore locations.

The uniform Hg, Pb, and Zn profiles at Station U-42 and the uniform Pb, Cu, and Zn profiles at Station B-8 are suggestive of mixing to a depth of 8 cm (Fig. 2 and 5). However, it is hard to explain the irregularity of the other elemental profiles at these locations if mixing is taking place. Kovacik and Walters (1973), however, show that mixing is only significant in the shallow nearshore waters of the Western basin on the basis of Hg profiles in the sediment. Skoch and Britt (1969) concluded that mixing occurred down to a depth of 5 cm at a site in the Western basin. Our own core data and the above cited data are inconclusive as to the extent of sediment mixing in the lake. Discussion of P-regeneration (Burns 1976) and sediment energetics (Thomas et al. 1976 and Lam and Jaquet 1976) show that turbulent mixing and sediment resuspension must occur in the shallow Erie waters. The nature and extent of surface sediment mixing in Lake Erie is poorly understood and requires further investigation.

NUTRIENT ELEMENTS

Included under this category are Org.-C, N, and P. The Nutrient elements make up the third largest group of materials in the recent sediments, accounting for up to 10% of the sediment by weight. Surface enrichment of these elements (Table 3, Fig. 2–7) has been previously ascribed to anthropogenic inputs of P to the lake with an ensuing increase in productivity (Kemp et al. 1974).

Concentration profiles for Org.-C in Lake Huron reflect small nutrient inputs and natural diagenesis of the organic matter (Kemp et al. 1972, 1974). Enrichments of Org.-C in Lake Huron are much less than those found in lakes Erie and Ontario and it can be assumed that the enrichments observed in Lake Erie are due exclusively to increasing organic loading. Nitrogen concentrations follow those of Org.-C because most of the sediment N is in an organic form (Kemp and Mudrochova 1972).

The statistical relationships observed between the Nutrient elements, Fe, Mn, Al, and the clay size fraction are summarized in the correlation matrix given in Table 8. As described by previous authors (e.g. Stangenberg 1949; Hutchinson and Vallentyne 1955; Tadajewski 1966 and Kemp 1971), Org.-C and N show an almost perfect relationship. The previously reported relationship of Org.-C to clay (e.g. Trask 1932; Van Straaten 1954; Bordovskiy 1965; Thomas et al 1972, 1973) is not seen. This lack of clay, Org.-C relationship is attributed to the increasing organic carbon loading to the sediment with time under conditions of increasing eutrophication and, as such, is independent of a relatively constant deposition of clay.

There is a good correlation between P and Org.-C (Table 8), reflecting the anthropogenic loading of P and commensurate increase in organic matter. There is also a reasonable correlation between P and Fe as noted by many previous workers (e.g. Williams et al. 1971).

In particular, Williams et al. (1976a) have concluded that there is no significant migration of P up the sediment column. Thus the P enrichment of the surface (0–1 cm) sediment is assumed here to be entirely due to increased loading of phosphorus to the lake. This assumption forms the basis for the estimates of anthropogenic loadings of P to the sediments which is discussed in the next section. The P loadings may be overestimated to the extent that this assumption is in error.

CARBONATE ELEMENTS

The Carbonate groupings of elements (Ca and CO_3-C and, as discussed previously, Mg) make up the second largest group in the sediments, accounting for up to 15% of the materials by weight. The SEF values (Table 4) show that Mg remains essentially constant with time. The Carbonate elements show a small surface enrichment in the central basin (Station G-16) and losses at the remaining stations (Table 4). The CO_3-C and Ca concentrations decrease at all stations from the base of the cores to the surface except at Station G-16 where these elements show a small increase (Fig. 2–7).

TABLE 8. Correlation coefficient matrix for Nutrient elements (74 samples).

	N	P	Fe	Mn	Al	Clay-Size fraction
Org.-C	.959**	.772**	.631**	.648**	.433**	.367**
N		.808**	.658**	.692**	.511**	.445**
P			.654**	.535**	.730**	.476**
Fe				.789**	.805**	.800**
Mn					.476**	.610**
Al						.754**

The Carbonate element distribution in the cores is poorly understood at the present time. Low carbonate concentrations in cores from Lake Superior are indicative of slow sedimentation and in situ dissolution of detrital carbonates (Dell 1973b). Presumably, the carbonates are presently undergoing in situ dissolution in the Erie surface sediments. X-ray diffraction of the surface sediment at these locations shows that calcite and dolomite are only present in minor amounts. The dolomite concentrations are a little larger, reflecting the greater solubility of calcite.

The linear correlation matrix for the Carbonate elements and the silt size fraction is given in Table 9. The coefficients indicate the presence of calcite (CO_3-C to Ca) and dolomite (CO_3-C to Mg), both showing a moderate relationship to the silt-size fraction. The dolomite and calcite relationship presumably reflects the higher concentrations of these minerals at the base of the cores. The carbonate–silt relationship is similar to that previously observed in Lake Huron and Ontario (Thomas et al. 1972, 1973) and for the surface sediments of Erie (Thomas et al. 1976). The carbonate cycle in Lake Erie appears to be interesting and, as is obvious from this discussion, requires much more detailed investigation.

TABLE 9. Correlation coefficient matrix for Carbonate elements (74 samples).

	Ca	Mg	Sr	Silt-size fraction
CO_3-C	.815**	.720**	−.060	.715**
Ca		.748**	.026	.565**
Mg			.209	.541**
Sr				−.157

MOBILE ELEMENTS

On the basis of the SEF values relative to the observed core profiles, Mn, Fe, and S are assigned to the Mobile group of elements. The term mobile, herein, is used to designate those elements which, due to variation in the physical–chemical properties of the core, are subject to dissolution and migration in the interstitial waters. These elements account for 5% of the total sediment by weight. Because of their mobility they can substantially influence the bottom waters of the lakes.

The vertical profiles of Mn in the cores can be broken down into two distinct groups: those that show a large increase at the sediment–water interface from a background level of around 0.1% or less and those that remain uniform with depth. Upward migration of soluble Mn under reducing conditions in sediment pore waters with surface precipitation by oxidation is well known (e.g. Mackereth 1966; Weiler 1973). This condition is best illustrated by the core profile for the eastern basin (Fig. 7). At this deepwater location where the hypolimnion waters are well oxygenated at all times, Mn is apparently concentrated in the surface oxidized sediment (0–1 cm). A similar but less enriched Mn concentration is observed at Station B-8 in the Central basin (Fig. 5).

Surface enrichment of Mn is not found at the Western basin locations (Fig. 2, 3, and 4). At these locations, it is believed that the seasonal redox conditions are such that overall precipitation of upward migrating Mn is not occurring and that the bulk of the Mn is being lost to the overlying waters. This situation is well illustrated by Station U-42 (Fig. 2). Anoxic conditions in the hypolimnion waters of the western and central basins (Beeton 1969; Burns and Ross 1972) account for the loss of Mn from the sediment interface at these locations. The high Mn concentrations at 1–2 cm at Station G-16 (Fig. 6) and the decrease at 0–1 cm in the same core is interesting. The profile is suggestive of ongoing release of Mn at the interface under the anoxic conditions which prevailed at the time this sample was collected. A more general account of Mn in the sediments of the Great Lakes has been present by Sly and Thomas (1974), who noted the sensitivity of both Mn and Fe to the degree of eutrophication in each lake.

The SEF values (Table 4) and the chemical profiles (Fig. 2–7) suggest that Fe might be better

placed with the Conservative elements. However, the studies of Burns and Ross (1972) have demonstrated the significance of the regeneration of soluble Fe from the sediments of the central basin under anoxic hypolimnion conditions, indicative of the mobile nature of this element.

Interpretation of the Fe profiles is difficult because the Fe concentration (avg = 4.5%) is much greater than that of Mn (avg = 0.12%). Hence any loss of Fe from the sediment–water interface to the hypolimnion waters equivalent to that of Mn will not be readily discernible. The statistical correlation between Fe and Mn ($r = 0.789$) and the general similarity of their profiles (Fig. 2–7) suggests that Fe is also being mobilized in the sediment. More detailed studies are required, particularly in determining the valence state of the Fe in order to make a more definite interpretation of Fe mobilization.

Total S increases below the sediment–water interface to a maximum value and then decreases again to lower values at the base of the cores (Fig. 2–7). The concentration profiles show a significant correlation with Eh ($r = -0.644$) and Org-C ($r = +0.693$). These correlations suggest that the bacterial oxidation of organic matter and subsequent decrease in Eh is accompanied by reduction of SO_4^{2-} and Fe^{3+} and formation of iron sulphides (Zobell 1946). Reduction of SO_4^{2-} has been observed in the surface sediments of the Central Basin after sedimentation of an algal bloom (Menon et al. 1971). Similar decreases in SO_4^{2-} concentration were found in a number of cores from the lake (Nriagu 1975). These findings, together with the color changes mentioned previously, indicate that fresh organic matter is being decomposed at the water interface with the formation of iron sulphides in the reduced zone of sediment. A more specific account of the S-cycle and diagenetic changes of S near the sediment–water interface in Lake Erie has been recently presented by Nriagu (1975).

Miscellaneous Elements

Be and V have been included under this category primarily due to the variability of their SEF values and irregular profiles. It is not certain whether these elements should be placed in the Conservative or Enriched grouping of elements. The enrichment of V in southern California sediments was demonstrated to be of anthropogenic origin (Bruland et al. 1974). The profiles for the two elements are also suggestive of enrichment. As the enrichment appears to be small, these elements are classed as Miscellaneous pending a more detailed assessment of the surficial sediment geochemistry.

Chronology and Sedimentation Rates

To interpret the recent sedimentary history of Lake Erie, it is necessary to establish a time-scale for sediment deposition. The increase in *Ambrosia* pollen (120 yr BP) and the decline in *Castanea* pollen (35 yr BP) provide the necessary chronologic framework for sediment studies (Anderson 1974; Kemp et al. 1974). The sediment, which lies above the *Castanea* horizon at each location, has been deposited since 1935 and is termed the recent sediment. Sediment located between the *Castanea* and *Ambrosia* horizons is termed the early-colonial sediment and represents materials deposited between 1850 and 1935. The pre-colonial sediment is that deposited prior to 1850.

In addition to interpreting the recent sediment history of Lake Erie, it is also possible to estimate averaged sedimentation rates for recent and early colonial sedimentation from the pollen horizons. The short-term averaged rates of deposition of fine-grained sediment will be variable, depending on water levels, storm activity, and the cultural activities in the drainage basin. Studies in Lake Michigan, utilizing ^{210}Pb dating, have shown that recent annual sedimentation rates have varied, from year to year (Robbins and Edgington 1975). In Lake Erie, where the sedimentation rates are much greater, these variations can be expected to be more extreme. Average sedimentation rates have been calculated for each sample location (Table 1). It should be emphasized that the rates are time-averaged and that the annual quantity of sediment being deposited at each of these locations could be much greater or much less depending on the climatic conditions. The sedimentation rates are expressed as g m^{-2} yr^{-1} and are calculated by dividing the dry weight per square meter of the sediment column between time horizons by the estimated time interval (Kemp et al. 1974).

The estimates show that the present-day sedimentation rates are very high in the Eastern and Western basins and are high in the Central basin (Table 1). The sedimentation rates are much greater than those found in the other Great Lakes and comparable large bodies of water (Kemp et al. 1974). The concentration of Hg in the Western basin sediments has been correlated with industrial inputs dating from 1939 and 1954, (Wolery and Walters 1974). The authors used pollen data from stations U-42, W-44, T-44, and B-8 to confirm the validity of the Hg horizons. Wolery and Walters (1974) and Walters and Herdendorf (1975) have constructed detailed distribution maps of the sedimentation rates in the Western basin, showing that the present-day rates are very high with values up to 1.5 cm yr^{-1}. The offshore

TABLE 10. Estimated quantity of silt and clay-sized sediment input from various sources.

Source	Zone	Yield of fine-grained material per annum (*Million metric tons*)
Shoreline erosion[a]	Detroit River — Point Pelee	0.4
	Point Pelee — Erieau	1.6
	Erieau — Long Point	21.2
	Long Point — Niagara River	0.2
	Detroit River — Maumee River	{<0.1}
	Maumee River — Sandusky	<0.1
	Sandusky — Ohio — Penn. Border	1.3
	Ohio — Penn. Border — Niagara River	{1.0}
	TOTAL	25.7
River inputs[b]	Detroit River	1.4
	Maumee River	1.8
	Other rivers	0.9
	TOTAL	4.1
Airborne particles[c]	Estimated range for the whole lake	0.2–3.3
Autochthonous organic matter[d]		1.0
Dredged spoils[e]	Whole lake	1.4

[a]Shoreline erosion data (Carter 1974; J. P. Coakley personal communication; W. S. Haras and J. Shaw personal communication). Bracketed quantities are our own estimates.
[b]Data from I.J.C. (1969).
[c]Whelpdale (1974 a, b).
[d]Decomposed residues of primary organic matter, see text for explanation.
[e]International Working Group Report (1974).

values are similar to our own estimates. The authors found a small zone off the mouth of the Detroit River where sedimentation rates were maximal for the basin.

We are continuing to study present-day sedimentation rates utilizing pollen chronology. The two high sedimentation rates in the Eastern basin are presently being confirmed by analysis of 10 further cores from the basin. Sedimentation rates 10 km to the north, east, and west of M-32 are of the same order of magnitude, while a location south of M-32 appears to have a lower rate. The Hg study (Wolery and Walters 1974) and the above results suggest that our present-day sedimentation rates are reasonably representative of the offshore locations.

It should be remembered throughout this discussion that the basin sedimentation rates have been calculated on the results of the close examination of only 10 cores. While we feel that these cores are well representative of their respective basins we are unable to estimate the accuracy of the calculated fine-grained sediment loads. This comment must necessarily apply to all the budgets calculated for the various elements (Table 13).

Sources of Lake Erie Fine-Grained Sediments

Modern Lake Erie sediment is derived mostly from shoreline erosion and river inputs of suspended solids. Secondary sediment sources include aeolian materials, autochthonous organic matter, subaqueous erosion, and offshore dumping of dredged spoils. There is little quantitative information on the amount of sediment entering Lake Erie from the various external sources. Using available data and making assumptions where the data is not available, we have estimated that the primary source of fine-grained sediment is from erosion of shoreline bluffs (Table 10).

The bulk (83%) of the sediment from shoreline erosion comes from the bluffs along the north shore of the Central basin between Erieau and Long Point (Table 10). It has been shown that erosion of these north shore bluffs is not only influenced by external factors such as large storms and lake level fluctuations but also by topographic and geologic factors (Quigley and Tutt 1968; Gelinas and Quigley 1973). The height and slope of the bluff, the shear strength properties of these materials and the erosional resistance properties

of these materials are the most important aspects. These bluffs, which consist of clayey-silt tills, range in height to 40 m above lake level and are eroding at rates of up to 3 m yr^{-1} (Gelinas and Quigley 1973). Estimates of the erosion of the northshore Canadian bluffs are based on the total quantity of silt- and clay-sized materials eroded over the last 4 yr (1970–74), which has been a period of high water levels (J. P. Coakley personal communication; W. S. Haras and J. Shaw personal communication). Therefore the bluff erosion estimates for the Canadian shoreline can be considered maximal. An averaged input of 21.2 million metric tons per year is calculated for this section of the north shore.

The sediment input from the remainder of the Canadian shoreline is much less (Table 10) because of the lower relief and geology of the shoreline. Clayey-silt bluffs with a maximum height of 30 m between Point Pelee and Erieau contribute 1.6 million tons. Intermittent bluffs along the shoreline between the Detroit River and Point Pelee have a maximum height of 14 m and contribute only 0.4 million tons of silt- and clay-sized material. Between Long Point and the Niagara River, the shoreline consists mainly of bedrock and sand dunes. As silty-clay tills are exposed only in a few bluffs, there is a small contribution of only 0.2 million tons to the lake (Table 10).

Inputs from erosion of the Ohio shoreline are based on the quantity of silt- and clay-sized materials eroded over the past 100 yr (C. H. Carter personal communication). The Ohio shoreline consists mainly of wetlands protected by barrier beaches between the Maumee River and Sandusky (Pincus 1960; Carter 1973). Although erosion rates of over 8 m yr^{-1} have been recorded in the past 100 yr for this shoreline (Benson 1974), the low relief and the small exposures yield an average of only 0.03 million tons of fine-grained sediment to the lake (Table 10). No data is available for the shoreline from the Detroit River to the Maumee River. As the shoreline in this region is similar to the above, it is assumed that the silt and clay contribution from this zone is less than 0.1 million tons.

The Ohio shoreline from Sandusky to the Pennsylvania border is variable with clayey-silt bluffs up to 20 m in height (Pincus 1960; Carter 1973). Bedrock also outcrops at the water level at several locations on this shoreline. Erosion rates of up to 2.5 m yr^{-1} have been recorded in Lake County (Carter 1974). The silt and clay contributions from this section of shoreline is estimated to be 1.3 million tons.

There is no information on erosion rates and bluff inputs for the south shore from the Pennsylvania border to the Niagara River. This shoreline consists of bluffs ranging in height from 1 to 30 m, composed of clayey-silt lacustrine sediments and tills. The base of the bluff at most locations consists of interbedded shales and siltstones. The bedrock assists in the protection of this shoreline from severe erosion like that experienced on the north shore. We estimate that less than 1 million tons of silt and clay sediment is contributed from this shoreline (Table 10).

River inputs of fine-grained materials total 4.1 million tons (I.J.C. 1969) with the Detroit and Maumee rivers making up to 80% of the totals (Table 10). The Maumee River has the largest river input into the lake, derived mainly from the soils of the Maumee drainage basin (Wall et al. 1974). The suspended materials in the Detroit River are mainly from Lake St. Clair and indirectly from southern Lake Huron (Duane 1967). Inputs from the Grand River, Ohio and the Grand River, Ontario, are small (0.4 million metric tons) relative to the Maumee and Detroit rivers. Although most of the sediment supply to Lake Erie is dominated by shore erosion in the Central basin, 80% of the sediment supply to the Western basin is from the Detroit and Maumee rivers. Therefore, the Western basin sediments are prone to show elevated levels of the Enriched elements shown in Table 3.

There is little information on the sources and quantity of airborne particulate matter over Lake Erie. An investigation of mass loading rates of particulate material in the southern Lake Huron–Lake St. Clair–western Lake Erie region over 13 mo, estimated residence times of between 5 and 16 h for the airborne particulate matter (Whelpdale 1974a). Using a simple model for the deposition of particulate matter into lakes, Whelpdale calculated deposition rates ranging from 0.2 to 3.9 $\mu g\ m^{-2}\ s^{-1}$. We calculated for Lake Erie that airborne particulate inputs would range from 0.2 to 3.3 million tons using this data (Table 10). It is probable that the airborne input would be in the lower part of the range, because of the dimensions of the lake and the short residence times.

Although the airborne input to the lake is less than that from shoreline erosion and river inputs, it is of great importance because the particles may have high concentrations of trace metals or nutrients. The observation of a considerable amount of anthropogenic materials in the surface sediments, mentioned previously, suggests that airborne particulate matter is likely to be important even at lake locations far removed from polluting sources.

Winchester and Nifong (1971) in a study of Lake Michigan proposed that aerosol fallout is

a major source of Zn, Cu, and Ni to the lake. Shiomi and Kuntz (1973) estimated that precipitation of nutrients to Lake Ontario contributed 12–14% of the total N-input and from 5–14% of the total P-input. Particulate fallout of Pb under moderate and low pollution conditions over the Great Lakes is estimated to range from 0.06 to 3.0 $\mu g\ cm^{-2}\ yr^{-1}$ of Pb (Whelpdale 1974b). Airborne loading to the lake will depend mainly on the meteorological conditions over the drainage basins. The prevailing westerly winds favor Detroit, Toledo and Cleveland rather than Buffalo as the major source cities for airborne pollution of the lake. Airborne materials as a source of anthropogenic loading to the sediments is further mentioned in the next section.

Autochthonous organic matter is also an important source of fine-grained sediment. Using the sedimentation rate data (Table 1), the Organic-C content of the offshore sediments (Kemp and Lewis 1968) and the assumption that 1/3 of the sediment organic matter is allochthonous (Kemp and Dell 1975), it is calculated that 1.0 million metric tons of organic matter are deposited and buried each year (Table 10). This quantity of autochthonous materials represents the residual primary organic matter after decomposition has taken place in the water column and at the sediment–water interface.

Dredging and subaqueous erosion are two other sources of fine-grained sediment. It is not possible to make any accurate estimate of the amount of silt- and clay-sized materials released to the lake from dredging activities. A volume of 2.3 million m^3 is dredged annually in Lake Erie for navigational purposes (International Working Group 1974). Assuming a water content of 75% by weight and a silt and clay content of 50% of all solids, dredging could yield around 1.4 million tons a year of fine-grained materials. If these assumptions are correct, dredging could be a significant source of fine-grained sediment.

Subaqueous erosion may be occurring on the cross-lake sills and in the nearshore zone (Lewis 1966; Rukavina and St. Jaques 1971 and Thomas et al. 1976). There is no estimate of the quantity of fine-grained sediment originating from this source that is redistributed within the lake.

Sinks of Fine-Grained Materials

Fine-grained materials that enter Lake Erie are either deposited on the lake bottom and buried or carried out of the lake via the Niagara River. Using our sedimentation rates from Table 1 for each basin of the lake, it is estimated that 29.6 million metric tons of fine-grained sediment are being deposited annually at the present time. The outflow of suspended particles in the Niagara River for 1968 amounted to 4.5 million metric tons (I.J.C. 1969). Assuming that 1.0 million tons of airborne particles are deposited in the lake annually, it is estimated that all external sources yield 33.2 million tons of sediment (Table 10). A sediment mass-balance shows a deficiency in source materials of 0.9 million tons:

Fine-grained sediment inputs (Table 10)	33.2
Outflow (I.J.C. 1969)	4.5
Net inputs	28.7
Sediment loading	29.6

The shoreline and river inputs and outputs are shown in Fig. 8.

At the present time there is not enough data to calculate the input of sand-sized materials to the lake. The sediment mass balance refers only to silt- and clay-sized materials. As a first attempt at a sediment budget for Lake Erie, it is surprising that such close agreement is found between the inputs and the outputs. Our estimates of shoreline erosion on the north shore are based on a 4-yr average and it would be expected that under times of lower water levels the input from this source would be much smaller. The sediment loading estimates are based solely on the average sedimentation rate (Table 1) and the area of fine-grained sediment of each basin (Fig. 8). These estimates yield a 35-yr average and are likely to be in error as sedimentation rates are variable within the lake. Similarly, our estimates of inputs from airborne particulates, dredged materials, and shoreline erosion of the Michigan, Pennsylvania, and New York shorelines require verification.

It is expected that as further information becomes available, the sediment budget will be drastically modified. However, several conclusions

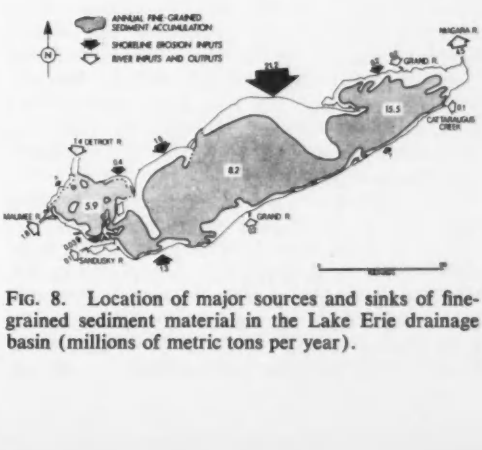

FIG. 8. Location of major sources and sinks of fine-grained sediment material in the Lake Erie drainage basin (millions of metric tons per year).

TABLE 11. Present-day natural and anthropogenic inputs of the Enriched and Nutrient elements for each sample location.

Station number	Loading	Enriched elements					Nutrient elements		
		Hg	Pb	Zn	Cd	Cu	Org.-C	N	P
			(*Loadings in* $\mu g\ cm^{-2}\ yr^{-1}$)						
				Western Basin					
U-42	A[a]	0.49	34.4	81.2	0.9	13.0	4460	645	105
	N[b]	0.02	10.7	31.8	0.6	9.4	5710	500	250
W-44	A	0.28	14.3	30.6	0.9	6.9	2870	450	30
	N	0.07	16.4	43.0	0.5	11.5	5410	415	320
T-44	A	0.06	5.4	10.9	0.2	2.6	1025	115	25
	N	0.004	1.6	6.5	0.04	1.9	925	95	50
				Central Basin					
B-8	A	0.07	6.8	12.4	0.1	2.9	1830	290	25
	N	0.007	2.7	8.9	0.2	2.6	1375	135	90
G-16	A	0.11	15.1	38.1	0.4	4.3	4495	635	100
	N	0.01	2.3	11.8	0.1	3.6	1610	210	140
				Eastern Basin					
M-32	A	0.15	44.3	113.1	1.2	12.7	10450	1520	305
	N	0.02	12.1	52.6	0.5	16.3	6220	760	405
Lake Ontario rainfall (Shiomi and Kuntz 1973)		—	1.4	5.3	0.9	0.4	—	100	4

[a]Anthropogenic loadings.
[b]Natural loadings

can be drawn from these results, which are important for an understanding of sediment movement in the lake:

1) Shoreline bluffs are the major source of natural sediment particles.
2) The north shore between Erieau and Long Point provides 60% of all incoming sediment.
3) Most of the fine-grained sediment is being deposited in the Eastern basin (52%).

Natural and anthropogenic inputs of the Enriched and Nutrient elements have been calculated for each sample location (Table 11) and for each basin (Table 12). The natural sediment inputs have been calculated by multiplying the present-day sedimentation rate by the Al-normalized surface (0–1 cm) concentration of each element. The anthropogenic input is then calculated by multiplying the difference between the total and normalized concentrations by the sedimentation rate (Bruland et al. 1974).

The results show that the anthropogenic loading is greatest at the Eastern basin location with the exception of Hg and Cu (Table 11). The latter elements are being deposited more rapidly in the Western basin. The sediment loadings are much greater than those reported for precipitation over Lake Ontario during 1970 and 1971 (Shiomi and Kuntz 1973). However particulates were not determined in the above study. Pb inputs of 3.0 $\mu g\ cm^{-2}\ yr^{-1}$ for airborne particulates under moderate pollution conditions (Whelpdale 1974b) are in the same range as the sediment Pb loadings (Table 11). There is not enough information at the moment to discuss the sources of the trace elements. The present evidence suggests that atmospheric fallout requires careful investigation.

As with the total sediment budget, anthropogenic loadings of the heavy metals and nutrients are greatest in the Eastern basin (Table 12). Although the major source areas for the anthropogenic materials are Detroit, Toledo, and Cleveland, the Eastern basin of the lake is acting as the major sink.

Comparison of our results with other estimates of Hg, N, and P inputs indicate that our values are of the right order of magnitude. The major Hg sources at Wyandotte, Mich. (4.5–9.1 kg/day) and Sarnia, Ont. (26.4 kg/day) are estimated to have released 3.3 and 9.6 metric tons, respectively, during 1970 (Wolery and Walters 1974). The Hg distribution in the surface sediments of the Great Lakes demonstrates the downstream movement of Hg from the source areas (Thomas 1974; Thomas and Jaquet 1976). Thus, our estimate of 16.5 metric tons of Hg in 1970 (Table 12) appears reasonable.

A materials balance for the lake, based on

TABLE 12. Total natural and anthropogenic loadings of the Enriched and Nutrient elements to the basins of fine-grained sediment in Lake Erie.

Area	Loading	Enriched elements					Nutrient elements		
		Hg	Pb	Zn	Cd	Cu			
		(*Metric tons per year*)					Org.-C	N	P
Western Basin	A	6.2	480	915	14	165	62,300	9,020	1,185
	N[b]	0.7	215	610	9	170	89,900	7,540	4,635
Central Basin	A	5.8	725	1,660	19	235	209,000	30,540	4,120
	N	0.5	160	680	9	200	98,500	11,410	7,585
Eastern Basin	A	4.5	1,350	3,440	37	385	318,000	46,140	9,290
	N	0.6	370	1,600	14	495	184,000	22,990	12,330
Whole lake	A	16.5	2,555	6,015	70	785	589,300	85,700	14,595
	N	1.8	745	2,890	32	865	337,700	41,940	24,550

[a]Anthropogenic loadings.
[b]Natural loadings.

1966–67 data, shows that 99,000 metric tons of N and 23,000 metric tons of P are retained within the lake (I.J.C. 1969). Our estimates of 127,000 tons of N and 39,000 tons of P (Table 14) are again of the right order of magnitude. Estimates of 28,000 metric tons (Burns 1976), which is mostly for anthropogenic P is also close to our own estimate for total P. It is not certain from the results that our breakdown into anthropogenic and natural P loading is correct. The P budget is further discussed in detail by Burns (1976), Lam and Jaquet (1976), and Williams et al. (1976a).

Dispersion, Deposition, and Burial of the Fine-Grained Sediments

The sedimentation estimates show that almost 30 million metric tons of fine-grained sediment are deposited annually, with 52% of this load accumulating in the downstream Eastern basin. The largest part of the natural sediment input is derived from erosion of the northern shore of the Central basin (Fig. 8). Obviously a large portion of the fine-grained materials from this source are being deposited and buried in the Eastern basin, 150 km away. Similarly, the anthropogenic loading estimates for the nutrient and trace elements (Table 11 and 12) show that the Eastern basin is the major sink for these materials. Although the sources of the anthropogenic elements are not well established, they presumably originate directly or indirectly from the large urban areas of Detroit, Cleveland, and Toledo. The downstream and upwind location of Buffalo would rule out this urban area as a major source of anthropogenic materials.

The natural and anthropogenic materials are obviously being transported large distances from their sources. There is little information on the nature of the transportation pathways and the length of time that these processes take. It is perhaps useful to speculate on the fate of a fine-grained sediment particle after its entry into Lake Erie.

Sediment particles upon entering the lacustrine system are subject to a variety of physical and chemical processes. Under quiescent conditions, the particles would settle to the lake bed at a rate dependent on their density, size, and shape. Settling rates of between -0.4 and 2.0 m day^{-1} were determined for seston in Lake Ontario (Burns and Pashley 1974). Quiescent conditions are rare and the particles are more likely to be transported away from their point of entry by the action of currents and eddy diffusion processes. The prevailing easterly direction of the currents (Simons 1976) will favor dispersion of the particles from the west towards the east. The distribution of Hg in the surface sediments clearly shows the dispersion and transport of this element (Thomas and Jaquet 1976). Eventually the particles will be deposited on the lake bed.

Fine-grained particles that are deposited in the high energy nearshore zones or on the cross-lake ridges (Thomas et al. 1976) are liable to be resuspended and further transported to zones of lower energy. It is thus possible for a sediment particle to be deposited and resuspended a great number of times, before reaching its final resting place in the lake. Herdendorf (1968) observed accumulations of up to 10 cm of sediment per year in the reef area of the Western basin, while areas close by received no accumulation. The author suggested that the trapped sediment was in transport across the area to a more quiescent zone. During this period of resuspension, the sediment particles may be subject to chemical or physical alteration and to incorporation into the

food chains. For example: a substance such as Hg may be recycled a great number of times before final deposition and burial (Wood 1974).

There are two pulses of high turbidity readings in Lake Erie. These occur in the spring and the fall (Powers et al. 1960), indicating that these are the periods when most resuspension will take place. Mean turbidity values in the fall are much greater than those in the spring (I.J.C. 1969). High turbidity values occur at all water depths, but higher values occur more frequently in the hypolimnion waters (Pinsak 1967; I.J.C. 1969; Burns 1976). The areal distribution of turbidity values indicate that the highest values occur in the Western basin, the north shore between Erieau and Long Point and on the south shore near Cleveland (I.J.C. 1969). The turbidity results suggest that resuspension is occurring in the bottom waters, particularly in the Western basin and during the fall. High values on the north shore, off Cleveland and in the Western basin are also related to shoreline erosion and the Maumee River inputs (Fig. 8).

Resuspension of the fine-grained sediment is the most likely mechanism for the transportation of the natural sediment. Anthropogenic materials may be transported also by other mechanisms. Aeolian sources may disperse particles directly over the lake to their zone of deposition. The observation of distinct suites of anthropogenic particles at each sample location (Table 2) suggests this possibility. Water quality and primary productivity data show that the highest concentration of nutrients and heavy metals and the greatest productivity occur in the Western basin and along the south shore of the Central basin (Weiler and Chawla 1968; Gachter et al. 1974; Glooschenko et al. 1974; Burns 1976a, b). The highest values are usually in the nearshore zones closest to the major population centers. It is possible that the nutrients and heavy metals are incorporated into the food chains and are transported from their nearshore source via low-density seston to the downstream regions of the lake. At this time it is also not certain whether the trace elements are associated with the autochthonous organic materials, the clay minerals, or the anthropogenic materials, discussed earlier.

On deposition, the particles will remain in the flocculent sediment–water interface zone, until burial to a depth of a few centimeters has taken place. Rapid alterations of the more labile materials will occur at the interface due to bacteria (Menon et al. 1971), bottom fauna, and chemical processes (Burns and Ross 1972). We have estimated that residence times will vary from 3 to 10 yr in the top 3 cm of sediment in the offshore areas of the lake. Substances are most likely to be transferred from the sediment to the bottom waters during this period. Under conditions of oxygen depletion, normally insoluble materials such as P, Mn, Fe, and S can also be transformed to the soluble state. Resuspension and mixing of the particles occurs in the flocculent zone due to the action of burrowing organisms, gas release, and turbulent mixing, as discussed previously.

It can be seen that little is known about the dispersion, deposition, and burial of fine-grained particles in Lake Erie. A knowledge of the transportation pathways is required. In particular, a time frame is necessary and estimates are needed for the residence times of the various chemical forms of the anthropogenic substances. Studies are also required on the fate of fine-grained particles after their deposition in the shallow offshore areas, such as in the Western basin, with emphasis on degree of mixing at the sediment–water interface and resuspension processes.

Summary and Conclusions

The offshore fine-grained sediments at six locations in Lake Erie exhibit generally similar physical characteristics. The sediments are composed of clay minerals, quartz, feldspars, carbonates, and organic matter. Distinct suites of anthropogenic particles (e.g. fly ash) are present at each location, with the greatest concentrations in the Central basin. Water contents, pH, and Eh decrease with depth of burial at most locations, generally becoming uniform below 30 cm. Positive Eh values are only found in the top few millimeters of sediment. The sediments are fine-grained, being made up of clay-size (50–75%) and silt-size (25–45%) particles.

In contrast to the physical characteristics, the chemical composition of the sediments is variable at each location. The chemical composition was normalized to the baseline Al concentration to correct for surface enrichments or depletions of carbonates and organic matter in the sediments. A sediment enrichment factor (SEF) was determined for each element, relating the excess, depletion or uniformity of elemental concentration to normalized baseline values. On the basis of the concentration profiles and the SEF values, the elements were broken down into six groupings: Conservative, Enriched, Nutrient, Carbonate, Mobile, and Miscellaneous elements.

The Conservative elements (Si, Al, K, Na, and Mg) are the most abundant in the sediment, indicative of the major mineralogical species. These elements have zero SEF values and their composition has remained essentially constant with

time, reflecting the unchanging terrigenous input from the land cover of the Lake Erie drainage basin.

The Enriched elements (Hg, Pb, Zn, Cd, and Cu) are characterized by positive SEF values and high concentrations above the *Ambrosia* horizon (~120 yr BP). The high surface concentrations of the heavy metals are due to anthropogenic loading of these elements in recent years. The Nutrient elements (Organic-C, N, and P) have similar characteristics to the Enriched elements, reflecting the increasing anthropogenic loading of P with commensurate increases in primary productivity. Baseline concentrations of Pb, Cu, Cd, and P in the sediment matched those of the northshore bluffs, which are the primary source materials.

The concentrations of the Carbonate elements (CO_3-C and Ca) generally increased from the sediment–water interface to the *Ambrosia* horizon. The lower surface carbonate concentrations are believed to be caused by dissolution of detrital carbonates in recent years.

The Mobile elements (Mn, Fe, and S) have chemical profiles that are related to the physical–chemical conditions in the sediments. At locations where the lake hypolimnion waters become seasonally anaerobic, linear Mn profiles are observed. In the Eastern basin where there are permanent aerobic conditions, Mn is concentrated in the surface oxidized zone. As with Mn, Fe concentrations suggest upward migration of the element in the pore waters and precipitation or partial release of the element in the oxidized zone. Total S concentrations increase as the Eh decreases, being greatest in the zone of most negative Eh where the S is held in the form of insoluble sulphides. The miscellaneous elements (Be and V) show no clear trend in their SEF values or concentration profiles. It is not clear whether these elements behave as Enriched or Conservative materials.

Erosion of shoreline bluffs is the major source of natural fine-grained sediment, contributing around 26 million metric tons of silt- and clay-sized materials each year to the lake. Bluff erosion between Erieau and Long Point on the Ontario shore yields 60% of the input from shoreline erosion. River inputs of suspended particulates are a secondary source of sediment, contributing only 4.1 million metric tons per year. Airborne particulates, dredged spoils, and autochthonous organic matter also contribute substantial quantities of sediment materials.

An annual loading of 30 million metric tons of fine-grained sediment is deposited on the lake bed, with 8 million tons accumulating in the Central basin and 15 million tons in the Eastern basin. These estimates show that the natural materials are transported a considerable distance from the north shore bluffs which are the major source. Natural and anthropogenic inputs of the Enriched and Nutrient elements parallel the sedimentation rates, with the greatest inputs to the Eastern and Western basins. Around 60% of the total heavy metal and nutrient loading to the sediments is deposited in the Eastern basin, indicating that the anthropogenic materials are also transported a long distance from the major source areas of Detroit, Toledo, and Cleveland.

The transportation pathways for the fine-grained sediment particles are discussed. The natural sediment is believed to be deposited and resuspended a great number of times before final deposition and burial in offshore areas. The fine-grained particles may also be subject to mixing at the sediment– water interface in the shallower zones. The anthropogenic heavy metal and nutrient inputs may be dispersed in three ways: 1) in association with clay minerals, the anthropogenic materials are subject to the same transportation processes as the natural particles, 2) after incorporation into the food chains, the heavy metals and nutrients can be biologically recycled and transported with low-density seston, and 3) some of the heavy metal input is associated with airborne particulates, which provide an alternative dispersion mechanism. It is not certain to what extent these three dispersion mechanisms operate.

Acknowledgments

We thank Dr C. H. Carter, Ohio Department of Natural Resources, Mr J. C. Coakley, Lakes Research Division, and Dr W. S. Haras and Mr J. Shaw, Shore Properties Studies Section, Canada Centre for Inland Waters, for their generosity in providing us with data on the volume of silt- and clay-sized materials eroded from the shoreline bluffs annually. Special thanks are also due to Mr A. Zeman for till porosity and bulk density data which enabled us to convert bluff volumes to mass. We would like to express our thanks to N. Harper for the pollen and carbon analyses and preparation of the data, A. Mudroch, L. Johnston, C. B. J. Gray, and L. Homer for assistance in sampling and sample preparation, T. W. Morton for major element analysis, M. Denford for heavy metal analysis, H. W. T. Wong for nitrogen analyses, and the officers and crew of the *Martin Karlsen*, *Limnos*, and *Lemoyne*. Finally, we would like to thank Dr P. G. Sly, Chief, Lakes Research Division, Canada Centre for Inland Waters for permission to publish this material.

ANDERSON, T. W. 1974. The chestnut pollen decline as a time horizon in lake sediments in eastern North America. Can. J. Earth Sci. 11: 678–685.

BEETON, A. M. 1965. Eutrophication of the St. Lawrence Great Lakes. Limnol. Oceanogr. 10: 240–254.

1969. Changes in the environment and biota of the Great Lakes, p. 150–187. *In* Eutrophication: causes,

consequences and correctives. Natl. Acad. Sci. Washington, D.C.

BENSON, D. J. 1974. Shore erosion along a low lying shoreline, western Lake Erie. Proc. Conf. Great Lakes Res. 17: p. 219.

BERNER, R. A. 1971. Principles of chemical sedimentology. McGraw-Hill, New York, N.Y. 240 p.

BORDOVSKIY, O. K. 1965. Organic matter in marine sediments. Mar. Geol. 3: 1–114.

BRINKHURST, R. O., A. L. HAMILTON, AND M. B. HERRINGTON. 1968. Components of bottom fauna of the St. Lawrence Great Lakes. Great Lakes Inst., Univ. Toronto, Toronto, Ont. Rep. P.R. 33: 50 p.

BRULAND, K. W., K. BERTINE, M. KOIDE, AND E. D. GOLDBERG. 1974. History of metal pollution in southern California coastal zone. Environ. Sci. Technol. 8: 425–432.

BURNS, N. M. 1976a. Oxygen depletion in the Central and Eastern basins of Lake Erie, 1970. J. Fish. Res. Board Can. 33: 512–519.

1976b. Nutrient budgets for Lake Erie, 1970. J. Fish. Res. Board Can. 33: 520–536.

BURNS, N. M., AND A. E. PASHLEY. 1974. In situ measurement of the settling velocity profile of particulate organic carbon in Lake Ontario. J. Fish. Res. Board Can. 31: 291–297.

BURNS, N. M., AND C. ROSS. 1972. Oxygen-nutrient relationships within the central basin of Lake Erie, p. 193–250. *In* H. E. Allen and J. R. Kramer [ed.]. Nutrients in natural waters. J. Wiley and sons, Inc., New York, N.Y.

CAMERON, E. M. 1967. A computer program for factor analysis of geochemical and other data. Geol. Surv. Can., Ottawa, Ont. Pap. 67–34: 42 p.

CARTER, C. H. 1973. Natural and man-made features affecting the Ohio shore of Lake Erie. Ohio Dep. Nat. Resour., Guideb. 1: 34 p.

1974. Evolution of a high relief Lake Erie Shoreline between 1876 and the present, Lake County, Ohio. Proc., Conf. Great Lakes Res. 17: p. 217–218.

CHAPMAN, L. J., AND D. F. PUTNAM. 1966. The physiography of southern Ontario. Univ. Toronto Press, Toronto, Ont. 386 p.

DELL, C. I. 1973a. A special mechanism for varve formation in a glacial lake. J. Sediment. Petrol. 43: 838–840.

1973b. Vivianite: An authigenic phosphate mineral in Great Lakes sediments. Proc. Conf. Great Lakes Res. 16: 1027–1028.

DOLCATER, D. L., J. K. SYERS, AND M. L. JACKSON. 1970. Titanium as free oxide and substituted forms in kaolinites and other soil minerals. Clays and Clay Miner. 18: 71–79.

DUANE, D. B. 1967. Characteristics of the sediment load in the St. Clair River. Proc. Conf. Great Lakes Res. 10: 115–132.

ERLENKEUSER, M., E. SUESS, AND H. WILLKOMM. 1974. Industrialization affects heavy metals and carbon isotope concentrations in recent Baltic Sea sediments. Geochim. Cosmochim. Acta. 38: 823–842.

GACHTER, R., R. A. VOLLENWEIDER, AND W. A. GLOOSCHENKO. 1974. Seasonal variations of temperature and nutrients in the surface waters of lakes Ontario and Erie. J. Fish. Res. Board Can. 31: 275–290.

GELINAS, P. J., AND R. M. QUIGLEY. 1973. The influence of geology on erosion rates along the north shore of Lake Erie. Proc. Conf. Great Lakes Res. 16: 421–430.

GLOOSCHENKO, W. A., J. E. MOORE, M. MUNAWAR, AND R. A. VOLLENWEIDER. 1974. Primary production in lakes Ontario and Erie: comparative study. J. Fish. Res. Board Can. 31: 253–263.

GOLDTHWAIT, R. P., A. DREIMANIS, J. L. FORSYTH, P. F. KARROW, AND G. W. WHITE. 1965. Pleistocene deposits of the Erie lobe, p. 85–97. In H. E. Wright Jr. and D. G. Frey [ed.] The Quaternary of the United States. Princeton Univ. Press, Princeton, N.J.

HERDENDORF, C. E. 1968. Sedimentation studies in the south shore reef area of western Lake Erie. Proc. Conf. Great Lakes Res. 11: 188–205.

HOUGH, J. L. 1958. Geology of the Great Lakes. Univ. Illinois Press, Urbana, Ill. 313 p.

HUTCHINSON, G. E., and J. R. W. VALLENTYNE. 1955. New approaches to the study of lake sediments. Verh. Int. Ver. Limnol. 12: 669–670.

INTERNATIONAL JOINT COMMISSION. 1969. Pollution of Lake Erie, Lake Ontario and the international section of the St. Lawrence River. Vol. 2 — Lake Erie. Int. Joint Comm. Rep.: 316 p.

INTERNATIONAL WORKING GROUP REPORT. 1974. Report on the abatement and control of pollution from dredging activites. Int. Joint Comm. Rep: 147 p.

ISKANDAR, I. K., and D. R. KEENEY. 1974. Concentration of heavy metals in sediment cores from selected Wisconsin lakes. Environ. Sci. Technol. 8: 165–170.

KEMP, A. L. W. 1971. Organic carbon and nitrogen in the surface sediments of Lakes Ontario, Erie and Huron. J. Sediment. Petrol. 41: 537–548.

KEMP, A. L. W., T. W. ANDERSON, R. L. THOMAS, AND A. MUDROCHOVA. 1974. Sedimentation rates and recent sediment history of Lakes Ontario, Erie and Huron. J. Sediment. Petrol. 44: 207–218.

KEMP, A. L. W., AND C. I. DELL. 1975. The geochemistry and mineralogy of Lakes Ontario and Erie bluffs, sediments, p. 50–58. *In* Proc. 3rd Annu. Conf. Environ. Earth Sci. Eng. Res. Rep. Seri. 20, Dep. Geol. Sci., Brock Univ., St. Catharines, Ont.

KEMP, A. L. W., C. B. J. GRAY, AND A. MUDROCHOVA. 1972. Changes in C, N, P and S in the last 140 years in three cores from Lakes Ontario, Erie and Huron, p. 251–279. *In* H. E. Allen and J. R. Kramer [ed.] Nutrients in natural waters. J. Wiley and Sons, Inc., New York, N.Y.

KEMP, A. L. W., AND C. F. M. LEWIS. 1968. A preliminary investigation of chlorophyll degradation products in the sediments of Lakes Erie and Ontario. Proc. Conf. Great Lakes Res. 11: 206–229.

KEMP, A. L. W., AND A. MUDROCHOVA. 1972. Distribution and forms of nitrogen in a Lake Ontario sediment core. Limnol. Oceanogr. 17: 855–867.

KEMP, A. L. W., H. A. SAVILLE, C. B. J. GRAY, AND A. MUDROCHOVA. 1971. A simple corer and a method for sampling the mud-water interface. Limnol. Oceanogr. 16: 689–694.

KOVACIK, T. L., AND L. J. WALTERS. 1973. Mercury distribution in sediment cores from western Lake Erie. Proc. Conf. Great Lakes Res. 16: 252–259.

LAM, D. C. L., AND J. M. JAQUET. 1976. Computations of physical transport and regeneration of phosphorus in Lake Erie, fall 1970. J. Fish. Res. Board Can. 33: 550–563.

LEE, G. F. 1970. Factors affecting the transfer of materials between water and sediments. Literature Rev. 1. Water Resour. Cent., Madison, Wis. 50 p.

LELAND, H. V., S. S. SHUKLA, AND N. F. SHIMP. 1973. Factors affecting distribution of lead and other trace elements in sediments of southern Lake Michigan, p. 89–129. *In* P. C. Singer [ed.] Trace metals and metal-organic interactions in natural waters. Ann Arbor Science, Ann Arbor, Mich.

LEWIS, C. F. M. 1966. Sedimentation studies of unconsolidated deposits in the Lake Erie basin. Unpubl. Ph.D. Thesis. Univ. Toronto, Toronto, Ont. 134 p.

MACKERETH, F. J. H. 1966. Some chemical observations on post-glacial lake sediments. Philos. Trans. R. Soc. London, Ser. B. 250: 165–213.

MALLARD, G. E., AND J. I. FREA. 1972. Methane production in Lake Erie sediments: Temperature and substrate effects. Proc. Conf. Great Lakes Res. 15: 87–93.

MANHEIM, F. T. 1970. The diffusion of ions in unconsolidated sediments. Earth Planet. Sci. Lett. 9: 307–310.

MENON, A. S., C. V. MARION, AND A. M. MILLER. 1971. Microbiological studies of oxygen depletion and nutrient regeneration processes in Lake Erie central basin. Proc. Conf. Great Lakes Res. 14: 768–780.

MORTIMER, C. H. 1971. Chemical exchanges between sediments and water in the Great Lakes — speculations on probable regulatory mechanisms. Limnol. Oceanogr. 16: 387–404.

MUROZUMI, M., T. J. CHOW, AND C. PATTERSON. 1969. Chemical concentrations of pollutant lead aerosols, terrestrial dust and sea salts in Greenland and Antarctic snow strata. Geochim. Cosmochim. Acta. 33: 1247–1292.

NRIAGU, J. O. 1975. Sulfur isotopic variations in relation to sulfur pollution of Lake Erie. *In* Isotope ratios as pollutant source and behavior indicators. IAEA Vienna. p. 77–93.

NRIAGU, J. O., AND C. I. DELL. 1974. Diagenetic formation of iron phosphates in recent lake sediments. Am. Mineral. 59: 934–946.

PINCUS, H. J. 1960. Engineering geology of the Ohio shoreline of Lake Erie. Ohio Div. Shore Erosion Tech. Dep. 7: 7 p.

PINSAK, A. P. 1967. Water transparency in Lake Erie. Proc. Conf. Great Lakes Res. 10: 309–321.

POWERS, C. F., D. L. JONES, P. C. MUNDINGER, AND J. C. AYRES. 1960. Applications of data collected along shore to conditions in Lake Erie. Proc. Conf. Great Lakes Res., Univ. Michigan, Great Lakes Res. Div., Publ. 5: 78 p.

QUIGLEY, R. M., AND D. B. TUTT. 1968. Stability, Lake Erie north shore bluffs. Proc. Conf. Great Lakes Res. 11: 230–238.

ROBBINS, J. A., AND D. N. EDGINGTON. 1975. Determination of recent sedimentation rates in Lake Michigan using Pb-210 and Cs-137. Geochim. Cosmochim. Acta. 39: 285–304.

RUKAVINA, N. A., AND D. A. ST. JACQUES. 1971. Lake Erie nearshore sediments, Fort Erie to Mohawk Point, Ontario. Proc. Conf. Great Lakes Res. 14: 387–393.

SHIMP, N. F., J. A. SCHLEICHER, R. R. RUCH, D. B. HECK, AND H. V. LELAND. 1971. Trace element and organic carbon accumulation in the most recent sediments of southern Lake Michigan. Environ. Geol. Notes, Ill. Geol. Surv., Urbana, Ill. 41: 25 p.

SHIOMI, M. T., AND K. W. KUNTZ. 1973. Great Lakes precipitation chemistry: Part I. Lake Ontario basin. Proc. Conf. Great Lakes Res. 16: 581–602.

SIMONS, T. J. 1976. Continuous dynamical computations of water transports in Lake Erie for 1970. J. Fish. Res. Board Can. 33: 371–384.

SKOCH, E. J., AND N. W. BRITT. 1969. Monthly variations in phosphate and related chemicals found in the sediments of the island area of Lake Erie, 1967–1968. Proc. Conf. Great Lakes Res. 12: 325–340.

SLY, P. G., and R. L. THOMAS. 1974. A review of geological research as it applies to an understanding of Great Lakes limnology. J. Fish. Res. Board Can. 31: 795–825.

STANGENBERG, M. 1949. Nitrogen and carbon in the bottom deposits of lakes and in the soils under carp-ponds. Verh. Int. Ver. Limnol. 10: 422–437.

TADAJEWSKI, A. 1966. Bottom sediments in different limnetic zones of a eutrophic lake. Ekol. Pol. — Ser. A, Tom XIV 21: 321–341.

THOMAS, R. L. 1969. A note on the relationship of grain size, clay content, quartz and organic carbon in some Lake Erie and Lake Ontario sediments. J. Sediment. Petrol. 39: 803–809.

1974. The distribution and transport of mercury in the sediments of the Laurentian Great Lakes system. Proceedings of the International Conference on transport of persistent chemicals in aquatic ecosystems. Ottawa, Ont. p. 1–16.

THOMAS, R. L., AND J.-M. JAQUET. 1976. Mercury in the surficial sediments of Lake Erie. J. Fish. Res. Board Can. 33: 404–412.

THOMAS, R. L., J.-M. JAQUET, A. L. W. KEMP, AND C. F. M. LEWIS. 1976. The surficial sediments of Lake Erie. J. Fish. Res. Board Can. 33: 385–403.

THOMAS, R. L., A. L. W. KEMP, AND C. F. M. LEWIS. 1972. Distribution, composition and characteristics of the surficial sediments of Lake Ontario. J. Sediment. Petrol. 42: 66–84.

1973. The surficial sediments of Lake Huron. Can. J. Earth Sci. 10: 226–271.

TRASK, P. D. 1932. Origin and environment of source beds of petroleum. Gulf Publishing Co., Houston, Tex. 323 p.

VANDERPOST, J. M. 1972. Bacterial and physical characteristics of Lake Ontario sediment during several months. Proc. Conf. Great Lakes Res. 15: 198–213.

VAN STRAATEN, L. M. J. V. 1954. Composition of recent marine sediments in the Netherlands. Leidse Geol. Meded. 19: 1–108.

WALL, G. J., L. P. WILDING, AND R. H. MILLER. 1974. Biological transformation of clay sized sediments in aquatic environments. Proc. Conf. Great Lakes Res. 17: 207–211.

WALTERS, L. J., AND C. E. HERDENDORF. 1975. Influence of the Detroit and Maumee Rivers on sediment supply and dispersal patterns in western Lake Erie. Proc. Conf. Great Lakes Res. 18.

WALTERS, L. J., C. E. HERDENDORF, L. J. CHARLESWORTH, H. K. ANDERS, W. B. JACKSON, E. J. SKOCH, D. K. WEBB, T. L. KOVACIK, AND C. S. SIKES. 1972. Mercury contamination and its relation to other physico-chemical parameters in the western basin of Lake Erie. Proc. Conf. Great Lakes Res. 15: 306–316.

WALTERS, L. J., T. J. WOLERY, AND R. D. MYSER. 1974. Occurrence of As, Cd, Co, Cr, Cu, Fe, Hg, Ni, Sb and Zn in Lake Erie sediments. Proc. Conf. Great Lakes Res. 17: 219–234.

WEILER, R. R. 1973. The interstitial water composition in the sediments of the Great Lakes. I. Lake Ontario. Limnol. Oceanogr. 18: 918–931.

WEILER, R. R., AND V. K. CHAWLA. 1968. The chemical composition of Lake Erie. Proc. Conf. Great Lakes Res. 11: 593–608.

WHELPDALE, D. M. 1974a. Particulate residence times. Water, Air, Soil Pollut. 4: 293–300.

1974b. Dry deposition over the Great Lakes. Proceedings of the Symposium on Atmosphere — surface exchange. Richland, Wash. (In press)

WILLIAMS, J. D. H., J.-M. JAQUET, AND R. L. THOMAS. 1976b. Forms of phosphorus in Lake Erie bottom materials. J. Fish. Res. Board Can. 33: 413–429.

WILLIAMS, J. D. H., AND T. MAYER. 1972. Effects of sediment diagenesis and regeneration of phosphorus with special reference to Lakes Erie and Ontario, p. 281–315. *In* H. E. Allen and J. R. Kramer [ed.] Nutrients in natural waters. J. Wiley and sons, Inc., New York, N.Y.

WILLIAMS, J. D. H., T. P. MURPHY, AND T. MAYER. 1976a. Forms and rates of accumulation of phosphorus in Lake Erie sediments. J. Fish. Res. Board Can. 33: 430–439.

WILLIAMS, J. D. H., J. K. SYERS, S. S. SHUKLA, AND R. F. HARRIS. 1971. Levels of inorganic and total phosphorus in lake sediments as related to other sediment parameters. Environ. Sci. Technol. 5: 1113–1120.

WINCHESTER, J. W., AND G. D. NIFONG. 1971. Water pollution in Lake Michigan by trace elements from pollution aerosol fallout. Water, Air, Soil Pollut. 1: 50–74.

WOLERY, T. J., AND L. J. WALTERS. 1974. Pollutant mercury and sedimentation in the western basin of Lake Erie. Proc. Conf. Great Lakes Res. 17: 235–249.

WOOD, J. M. 1974. Biological cycles for toxic elements in the environment. Science 183: 1049–1052.

ZOBELL, C. E. 1946. Studies on redox potential of marine sediments. Am. Assoc. Pet. Geol. Bull. 30: 447–513.

Forms of Iron and Manganese in Lake Erie Waters

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BURNS, N. M., AND J. O. NRIAGU. 1976. Forms of iron and manganese in Lake Erie waters. J. Fish. Res. Board Can. 33: 463–470.

About 95% of the iron observed in the lake water is bound to inorganic particles and is present in the water because of sediment resuspension. Contrary to popular belief, the upwelling of interstitial waters charged with ferrous iron does not appear to play an important part in the regeneration of iron from Lake Erie sediments except when anoxic conditions exist at the sediment–water interface. By contrast, manganese is usually found in the overlying water as a result of the release of interstitial water containing dissolved manganese. This precipitates in oxygenated water so that most of the manganese present in the water is found in the particulate form. However, the released manganese will remain soluble if the oxygen saturation is less than 50%. In general, it was found that both the iron and manganese were in the particulate or soluble form in accordance with the geochemical solubility rules.

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Environ 95% du fer observé dans les eaux du lac est fixé à des particules inorganiques, et sa présence dans l'eau est le résultat d'une resuspension du sédiment. Contrairement à la croyance populaire, la remontée d'eaux interstitielles chargées de fer ferreux ne semble pas jouer un rôle important dans la régénération du fer des sédiments du lac Erié, sauf en présence de conditions anoxiques à l'interface sédiment–eau. Par contraste, le manganèse se trouve ordinairement dans l'eau qui recouvre le sédiment par suite de la mise en liberté d'eau interstitielle contenant du manganèse dissous. Il précipite dans l'eau oxygénée, de sorte que la plus grande partie du manganèse présent dans l'eau se trouve sous forme particulaire. Cependant, le manganèse libéré demeurera soluble si la saturation d'oxygène est inférieure à 50%. En général, on constate que le fer et le manganèse sont deux sous la forme particulaire ou soluble suivant les règles de solubilité géochimique.

Accepted November 17, 1975

Accepté le 17 novembre 1975

THE cycling of iron and manganese within and through aquatic ecosystems has been extensively investigated because of the synergistic effects on the other key elemental cycles. Thus, the precipitation and dissolution of iron and manganese oxyhydroxides in response to changes in pH and Eh are believed to be an important buffer mechanism regulating the levels of phosphorus and trace metals in aquatic environments (Mortimer 1941, 1942; Hutchinson 1957). Furthermore, several recent papers (Davies 1970; Lewin and Chen 1973; Murphy and Lean 1975) have suggested that iron and/or manganese may on occasion be the limiting element(s) for phytoplankton growth in some lacustrine environments.

The literature on the behavior of Fe and Mn in abiotic, well-defined systems is voluminous (e.g. see Garrels and Christ 1965; Stumm and Morgan 1970). The data from these systems are frequently used to develop theoretical models predicting the geochemical behavior of Fe and Mn in aquatic environments. These chemical models presume equilibrium (or steady-state) conditions and that the effects of organisms on the water chemistry are indirect — notably in mediating the changes in pH, Eh, and concentration of organic chelators. In spite of the obvious limitations, these models have been shown to approximate fairly closely the observed behavior of Fe and Mn in many lakes (see Delfino and Lee 1968; McMahon 1969). An objective of this report will be to relate the chemistry of Fe and Mn in Lake Erie to the published theoretical models.

It is widely recognized that manganic oxyhydroxides are more easily reduced than the ferric oxyhydroxides under naturally occurring pH–Eh conditions (see Krauskopf 1957). As a lake becomes progressively depleted in oxygen, one would, therefore, expect the differential dissolution of the Mn and Fe oxyhydroxides (in sediments and particulate materials) to result initially in a higher concentration of dissolved

Printed in Canada (J3998)
Imprimé au Canada (J3998)

Mn than dissolved Fe in the lake water. This report also examines the variations in the concentrations of dissolved Mn and Fe induced by changes in the redox potentials of the lake. The mechanism for the geochemical separation of Mn from Fe is of interest in view of the recent theories (Cronan and Thomas 1972; Rossmann and Callender 1968) on the origin of ferromanganese nodules in the Great Lakes.

Methodology

The locations for the sample stations are shown in Fig. 1. The six stations were surveyed seven times between May and November 1971, at monthly intervals. The water samples were taken by means of a submersible pump. The sampling depths were chosen so that samples were taken just above, within, and just below the thermocline with two other samples taken at equally spaced depths in the epilimnion and another in the hypolimnion, if possible.

Immediately after collection, the samples were filtered through precombusted and preweighed GF/C filter papers. On returning to shore, these papers were combusted and weighed so that the quantity of suspended inorganic material (SIM) in the water could be calculated. The rather coarse GF/C (porosity range, 0.6–1.2 μ) filter papers were used to prevent excessive filtering times, which could lead to significant oxidation of the dissolved iron and manganese species (see e.g. Lewis and Chen 1973). The filtrate was quickly acidified (with HCl) and subsequently analyzed in the laboratory according to the method of Traversy (1971). Total iron and manganese in the corresponding acidified but unfiltered water samples were also determined in the shore laboratory. The difference between the iron concentrations in the unfiltered and filtered water samples is here referred to as particulate iron $(Fe)_p$, while the iron in the unfiltered water is referred to as total iron $(Fe)_t$ and, in the filtered water, as soluble iron $(Fe)_s$. Similar designations are made for the corresponding manganese components. The term "soluble" will be used here even though it may be misleading since some of the iron passing through the GF/C filter may be associated with particulates. Regression analysis was done on parameter values measured from the same sample rather than by comparing average hypolimnion or epilimnion station values.

Measurements of Eh and pH were made on unfiltered water samples soon after sample retrieval. A Corning pH meter was used for the pH measurements which are estimated to be correct to 0.1 pH units. Zobell's solution was used to calibrate the platinum electrode and the Eh values are estimated to have a precision of 20 mv.

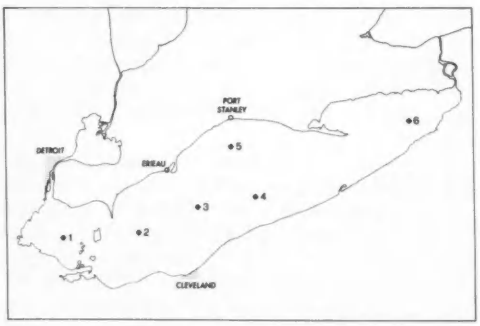

FIG. 1. Lake Erie sample stations.

Results

An example of the data that were collected at each station is shown in Fig. 2. These data were collected at Station 3 toward the end of the stratified period when the differences between the hypolimnion and epilimnion values were at their greatest. For complete data, see Burns et al. (1976). It should be noted that, although occasionally, low oxygen concentration (5.0% O_2 saturation) was observed in the water, anoxic water was not encountered during this study.

The regression analyses showed significant correlations between $(Fe)_t$ and SIM and $(Mn)_t$ and SIM under both isothermal and stratified conditions. The numerical results of the analyses are shown in Table 1 and a graphical display of the data is shown in Fig. 3 (A–D). In calculating the regression relationship shown in Fig. 3B, the high value for $(Fe)_t$ (of 16.7 μM Fe liter^{-1} shown as a solid triangle) was omitted since the

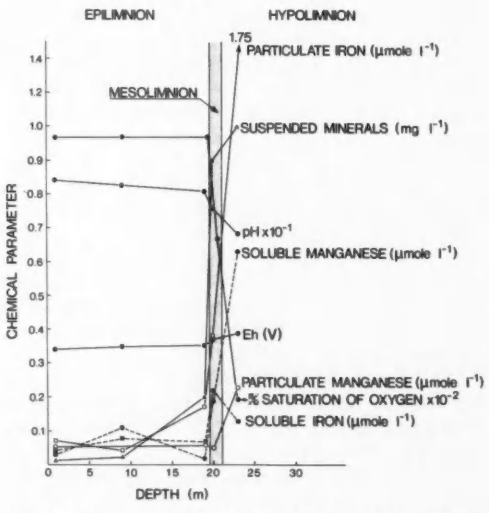

FIG. 2. Observed chemical variables at Station 3 on September 1, 1971.

TABLE 1. Regression equations for $(Fe)_t$ and $(Mn)_t$ as related to SIM concentrations in Lake Erie waters. (Minimum number of data pairs is 42; correlation coefficient is significant at the 99% level when greater than 0.39).

Equation no.	Fig. no.	Thermal condition	Equation of Line (Units: slope = μM liter^{-1} per mg liter^{-1} : constant = μM liter^{-1})	Correlation coefficient
1	3A	isothermal	$(Fe)_t = (0.71 \pm 0.0)$ SIM $+ (0.0 \pm 0.0)$	0.88
2	3B	stratified	$(Fe)_t = (0.50 \pm 0.11)$ SIM $+ (0.16 \pm 0.6)$	0.55
3	3C	isothermal	$(Mn)_t = (0.011 \pm 0.005)$ SIM $+ (0.079 \pm 0.023)$	0.46
4	3D	stratified	$(Mn)_t = (0.60 \pm 0.002)$ SIM $+ (0.001 \pm 0.002)$	0.82

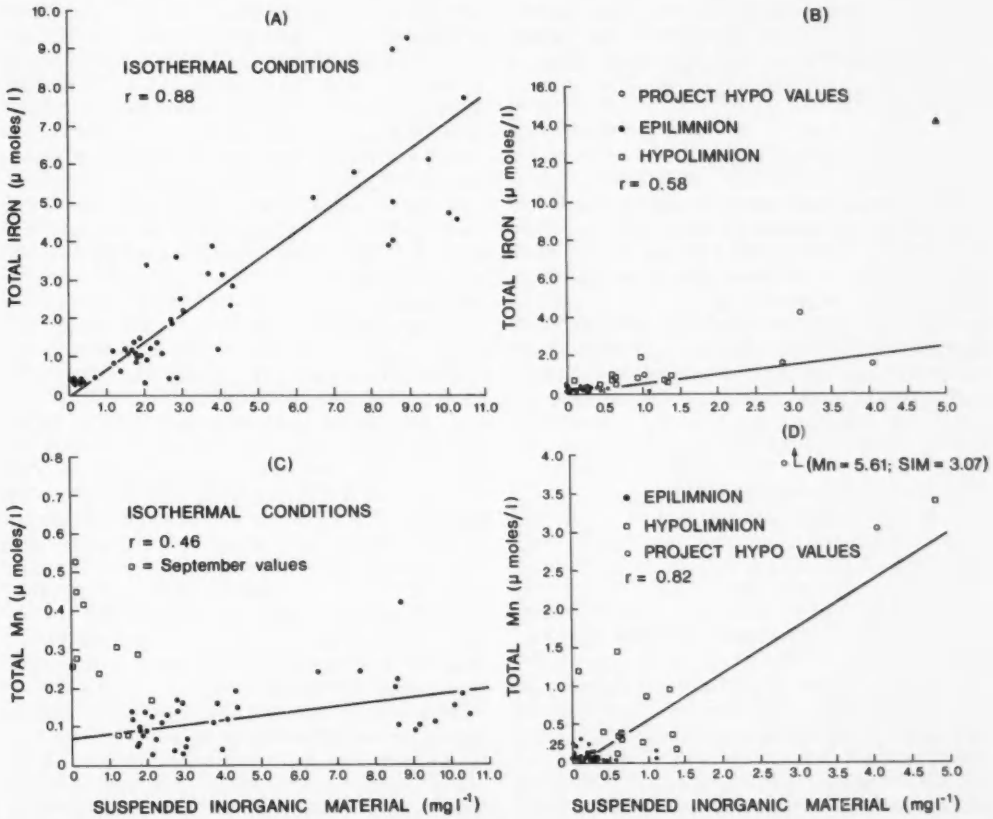

FIG. 3. Total iron and suspended inorganic material of samples during A) unstratified conditions and B) stratified conditions (value observed for Stn. 2 on Aug. 31, 1971 is shown by ▲.) and total manganese and suspended inorganic material samples during C) unstratified conditions and D) stratified conditions. The Project Hypo values (○) were not included in the correlation calculation for (B) and (D).

value was considered to be atypical for reasons discussed below. Values of total iron measured during the previous year under oxygenated conditions (Project Hypo values; Burns and Ross 1972) fall near the computed regression line and illustrate the abnormality of high 1971 value for $(Fe)_t$. On the other hand, the high concentration of $(Mn)_t$ observed in 1971 is considered normal since values of similar magnitude were observed during the previous year under oxygenated con-

ditions (Project Hypo values) and was included in the regression calculation.

When the data for the particulate and dissolved forms of Mn and Fe are plotted (Fig. 4), it can be seen that almost all of the iron in the samples is in the particulate form. Some of the samples having a low $(Fe)_t$ content do show a high proportion of $(Fe)_s$ relative to $(Fe)_p$ content. However, this may be due largely to the leakage of fine particulate matter through the somewhat coarse filter papers used in this investigation. There were no samples with an $(Fe)_t$ concentration of greater than 1.0 μM liter^{-1} which contained more than 10% of $(Fe)_s$. For this reason, it can be stated that in general the $(Fe)_p$–SIM relationships are very similar to those demonstrated by the $(Fe)_t$–SIM correlations.

Discussion

The results of the analyses on the samples taken under isothermal conditions show that iron constitutes about 4.0% of the SIM, on average (see Fig. 3A and 4). The data of Kemp et al. (1976) show that iron forms a fairly constant fraction of from 3% to 5% of the fine-grained surficial sediments of Lake Erie. The close correspondence in the proportion of iron in the sediments and in the SIM leads to the conclusion that the iron in unstratified water in Lake Erie is in association with the suspended materials. The particulates are probably derived by the resuspension of bottom materials within the lake and as erosional loadings to the lake. It is unlikely that a major part of the suspended iron is the result of river inflows because similar ratios of $(Fe)_t$ to SIM were observed at stations both close to and far from river mouths. The study of Lam and Jaquet (1976) demonstrates the ease and frequency of the resuspension of bottom materials in Lake Erie. Furthermore, these findings are in agreement with those of Carroll (1958) who found that the iron remained stably associated with the clays in oxygenated waters of pH above 7.0. However, there is considerable erosional input to the Central Basin (Kemp et al. 1976) and this would result in some fine clays being suspended in the waters of this basin which apparently have the same $(Fe)_t$: SIM ratio as the sediments.

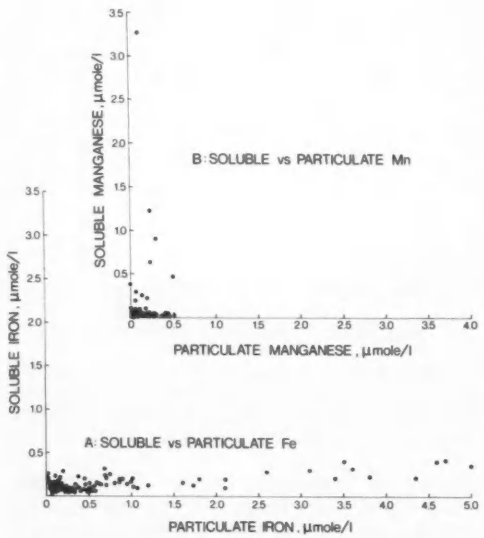

FIG. 4. Particulate and soluble A) iron and B) manganese concentrations in Lake Erie waters.

It also appears that most of the iron is associated with the suspended materials under conditions of thermal stratification, with the $(Fe)_t$ equal to 2.8 ± 0.6% of the SIM on average (see Fig. 3B). The reason for the somewhat different proportions of iron within the SIM under the two thermal conditions is not certain. The suspended inorganic material in a lake consists largely of resuspended clay but does also incorporate the siliceous frustules resulting from diatom growth. Under conditions of thermal stratification, the concentration of suspended clays in the hypolimnion was low and in these circumstances the concentration of iron-free diatom frustules may be large enough to dilute the $(Fe)_t$ content of the SIM. The similarity between the proportion of iron in SIM under oxygenated conditions and iron in the sediments is marked and it is suggested that the correlation between $(Fe)_t$ and SIM is the result of a causal relationship; iron in oxygenated waters of Lake Erie is largely due to the presence of suspended and resuspended inorganic materials in the water.

At station 2 (Aug. 31, 1971), the $(Fe)_t$ within the water was found to be equivalent to 16.2% of the SIM even when the water carried a high concentration of SIM (4.86 mg liter^{-1}). The hypolimnion water had an Eh value of −40 which would imply a considerably lower Eh at the mud–water interface. The low Eh value suggested the regeneration of iron in soluble form out of the sediments. The $(Fe)_t$ consisted of 95% $(Fe)_p$ and 5% $(Fe)_s$ and it is probable that the iron dissolved out of the sediments under the low Eh conditions and was then being precipitated in the overlying water containing oxygen at 5% of saturation. The reprecipitated iron remained in suspension for some time, accounting for an

unusually large concentration of Fe in the SIM. The observed low concentration of oxygen was evidently sufficient to cause the precipitation of the iron. During Project Hypo (Burns and Ross 1972), unusually high concentrations of either $(Fe)_t$ or soluble reactive phosphorus were never observed if there was even 0.1 mgm O_2 liter^{-1} (approx. 1.0% saturation) in the water. The observations lead to the conclusion that water in Lake Erie must be devoid of oxygen for soluble iron to be present at concentrations above 0.3 μM liter^{-1}. High concentrations of soluble iron (3.0 μM Fe·liter^{-1}) were observed in anoxic water in Lake Erie in 1970 during Project Hypo.

The data in Fig. 3C show that the $(Mn)_t$:SIM ratio is not constant as in the case of the $(Fe)_t$:SIM ratio (see Fig. 3A). Manganese constitutes approximately 0.12% by weight of the fine-grained sediments (Kemp et al. 1976). It can be deduced from equation 3 (Table 1) that the Mn:SIM ratio in the suspended particles is less than the Mn:SIM ratio for the sediments at concentrations of more than 9.0 mg SIM liter^{-1} in the overlying water. However, the Mn:SIM ratio for the particles in the water is usually greater than the same ratio for the sediments at SIM concentrations of less than 9.0 mg SIM liter^{-1} considering the intercept value of 0.08 μM Mn liter^{-1} (equation 3, Table 1). Since SIM concentrations of greater than 9.0 mg SIM liter^{-1} are seldom encountered, it can be concluded that the particles in the lake water are nearly always enriched in manganese when compared with the sediments.

It is probable that a large fraction of the Mn in the lake water is derived from the sediments by the outflux of the interstitial water to the overlying water. This is possible since the concentration of Mn in the interstitial water of surface sediments (0–2 cm) is high (13–130 μ moles Mn liter^{-1}; Nriagu and Dell 1974) relative to that in the lake water. The regeneration of the Mn from sediments has also been suggested by Kemp et al. (1976) to explain their observations. Since there is a positive correlation between the $(Mn)_t$ and SIM, it is probable that the water turbulence which resuspended the sediments also caused the extraction of the interstitial water containing soluble manganese from the sediments. However, the weak correlation between $(Mn)_t$ and SIM is not considered to illustrate a cause and effect relationship between $(Mn)_t$ and SIM, but rather demonstrates that increased concentrations of both are caused by the same process, probably the disturbance of the sediments by bioturbation or water turbulence. The manganese in the hypolimnion waters during August was found to be predominantly in the soluble form. In September, after the complete overturn in the Central basin, unusually high values of $(Mn)_t$ were observed and the data are shown in Fig. 3C. The manganese in these samples was predominantly in the particulate form and was derived from the high concentrations of soluble manganese in the hypolimnion waters just prior to the overturn. The concentrations of $(Mn)_t$ observed during the following month, October, at the same Central basin stations were considerably lower than those observed during September. The experimental studies of Delfino and Lee (1968) have also shown a similar conversion of soluble manganese to particulate manganese and the subsequent slow precipitation of the $(Mn)_p$ from anoxic hypolimnion water after reoxygenation. It would then appear that a similar process occurred during this investigation.

The most obvious feature of the data shown in Fig. 3D is that the hypolimnion concentrations of $(Mn)_t$ were much higher than those observed for the epilimnion waters during the same period. Since the sediments appear to be the prime source of manganese, it is to be expected that the hypolimnion waters in contact with the sediments would contain more manganese than the epilimnion waters. Also, it was found that the ratio of $(Mn)_s$:$(Mn)_p$ changed from less than 1.0 to 2.0 at oxygen concentrations of less than 50% saturation and at these lower oxygen concentrations, the concentration of $(Mn)_s$ was more than 0.2 μM liter^{-1} (see Fig. 4 and Burns et al. 1976). Delfino and Lee (1968) found the conversion of soluble Mn into particulate Mn took about 6 h when oxygen was introduced into anoxic water. Thus, the $(Mn)_s$ observed in this investigation probably was regenerated from the sediments in soluble form and remained soluble in the water containing less than 50% saturation of oxygen.

The slope in the line in Fig. 3D indicates that the hypolimnion water can be enriched such that the $(Mn)_t$:SIM ratio in the water was 30 times larger than the same ratio for the sediments. This ratio is considerably higher than the manganese enrichment of the unstratified water with oxygen concentrations of close to 100% saturation. The lower enrichment of Mn in well-oxygenated water is probably due to the regenerated soluble manganese precipitating and settling out of the water column.

The Eh and pH conditions of the lake water during the cruises are plotted in Fig. 5. It is seen from Fig. 5A that the limnological behavior of manganese conforms largely to the predicted behavior based on equilibrium phase diagrams (see e.g. Stumm and Morgan 1970). There were

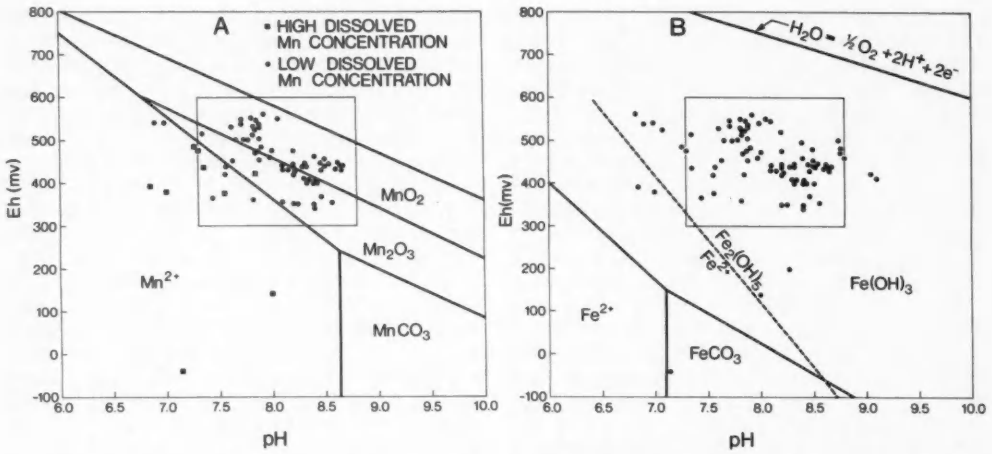

FIG. 5. A, Eh–pH conditions of samples taken during the study in relation to the stabilities of Mn minerals. Field boundaries are for activities of dissolved species of 10^{-6}. Thermochemical constants for the solid phases obtained from Stumm and Morgan (1970). The predominant Eh, pH conditions for the lake water are enclosed in the box; B, Stabilities of iron minerals in relation to the Eh–pH conditions of Lake Erie waters. Field boundaries and source of thermochemical constants same as for Fig. 5A. Free energy value for $Fe_2(OH)_5$ from Nriagu and Dell (1974).

10 samples taken which had $(Mn)_s$ levels greater than 0.2 μM liter^{-1}; this may be considered sufficient evidence for the assumption that there was a significant concentration of Mn^{2+} ions. It was found that eight out of these 10 samples had Eh and pH conditions which fell in the Mn^{2+} field in the Eh–pH diagram for manganese oxyhydroxides. There were some samples which apparently did not contain a significant concentration of Mn^{2+} ions which, nevertheless, fell in the Mn^{2+} field. However, there was a 90% conformity to the Eh–pH diagram among the samples which were in the solid phase fields.

All the Eh, pH data plot within the $Fe(OH)_3$ stability field (Fig. 5B). The displacement of the measurements from the Fe^{2+} – Fe $(OH)_3$ field boundary would seem to be anomalous, remembering that a small fraction of the iron may be in soluble form. It is, however, possible that the (dissolved) iron is bound to soluble compounds and hence out of equilibrium with the solid oxyhydroxide phase. Another possible explanation for the displacement is that the dominant solid phase consists of ferrosoferric hydroxide. Fig. 5B shows that the formation of $Fe_2(OH)_5$ (used here only as an example) would result in some of the Eh, pH measurements falling in the stability field of Fe^{2+}. A characterization of the nature of iron in the particulate material would be of interest.

The differential response of Fe and Mn to changes in Eh and pH of the lake water is illustrated in Fig. 2. It was found that 50% depletion of oxygen saturation in the lake water is sufficient to engender a large scale dissolution of solid Mn phases. By contrast, the dissolution of the solid Fe phases does not occur when the water masses contain even 1% saturation of oxygen. A study of the dissolution of freshwater ferromanganese oxides under controlled Eh and pH conditions (Nriagu 1974) in fact shows that the dissolution of iron oxides does not commence until the Mn oxides are completely removed from the mixed oxides.

Because of the leakage of fine particulates through the somewhat coarse filter papers used in this investigation, it is not possible to determine accurately the concentration of the truly dissolved species. However, it has been demonstrated that there is a fair agreement between the thermodynamically predicted and the observed behavior of iron in Lake Erie. If indeed the ferric oxyhydroxides control the iron concentration, the total concentration of dissolved iron in the oxygenated waters of Lake Erie may be expected to be 10^{-8} molar. In other words, the concentration of dissolved iron is likely to be almost negligibly small. The manganese hydroxide solubility is not so readily determined because of the possible changes in the nature of the solid phases. However, three lake-wide cruises in Lake Erie during 1970 established that surface water which was

filtered through 0.45-μ filters contained less than .02 μM Mn liter^{-1} (Burns et al. 1976). Thus even while the concentrations of total iron and manganese is the water may be relatively high, the iron would be strongly associated with clay while the manganese would be present in a precipitated form and both may thus be biologically unavailable. Since the equilibrium concentrations of Fe^{3+} and Mn^{3+} are so low it is possible that phytoplankton could be starved of iron or manganese even though the total concentrations of the elements may still be high. This situation could easily exist in the deep well-oxygenated parts of the Great Lakes unless there is an adequate supply of organic chelating compounds in the water which are also assimilable by the phytoplankton. These compounds could result in an adequate concentration of soluble iron and manganese. It is of considerable interest to know the nature and concentration of the dissolved iron and manganese in the Great Lakes, especially in light of the need of iron by blue-green algae during nitrogen fixation (Murphy and Lean 1975) and the observed correlation of the distribution of soluble manganese with chlorophyll *a* observed in Lake Ontario (Chau et al. 1970).

Summary and Conclusions

The study of iron and manganese in lakes is of interest both from the geochemical and biological point of view. Geochemically, one is interested in learning whether the solubility models developed for completely inorganic systems apply to lake systems containing high concentrations of organic materials. Biologically, iron and manganese are essential trace elements which can be depleted to limiting levels; thus a knowledge of the iron and manganese available to phytoplankton is important.

Samples were analyzed for particulate iron and manganese, soluble iron and manganese, oxygen concentrations, pH, Eh, and for suspended inorganic material (SIM). Correlation calculations on the results of samples from oxygenated water revealed a definite proportionality of total iron to SIM which was very similar to the proportion of iron found in the underlying sediments. This led to the conclusion that the iron observed in Lake Erie waters is primarily a result of sediment resuspension with the iron being strongly bound to the clays. The observed Fe:SIM proportionality definitely indicates that the iron observed in the water was not in the form of iron oxides precipitated from upwelling interstitial water but was attached to resuspended clays. There was the possible release of iron dissolved in interstitial water in a situation where anoxic conditions had existed at the sediment–water interface for some time. This situation can exist in the Central basin toward the end of the stratified season when large amounts of iron can be dissolved from the clays during anoxic conditions. This iron is precipitated from the water when it is reoxygenated and settles onto the sediments and will contribute to the enrichment of the surface sediments of the Central basin in iron as observed by Kemp et al. (1976).

Manganese is seen to behave quite differently to iron in that the particles in the waters of Lake Erie were found to be relatively enriched in manganese in comparison with the sediments. Soluble manganese is extracted from the sediments by physical processes and is slowly reprecipitated at the sediment interface where the pore waters encounter oxygenated lake waters. In hypolimnetic waters with oxygen concentrations less than 50% of saturation, the reprecipitation is slower than the regeneration from the sediments because the manganese concentration in these waters increases with time. This regenerated material reprecipitates at the time of the fall overturn. Manganese thus can be transported independently of the clay materials in Lake Erie and since the water in the lake travels from west to east, the manganese extracted from the sediments in the Western basin could be expected to settle finally in the Eastern basin. This transport process may be part of the reason for the clay materials in the Western basin having about half of the content of manganese of the sediment in the Eastern basin (Kemp et al. 1976).

Acknowledgments

We would like to thank Mr F. Rosa for his help with the field work and sample analysis. His assistance with the data analysis is gratefully acknowledged.

BURNS, N. M. AND C. ROSS. 1972. Oxygen-nutrient relationships within the Central Basin of Lake Erie, p. 85–119. *In* N. M. Burns and C. Ross [ed.]. "Project Hypo." Can. Cent. Inland Waters Pap. 6.

BURNS, N. M., F. ROSA, AND C. H. CHAN. 1976. Chemical parameters observed in Lake Erie 1970–1971. Can. Cent. Inland Waters Pap. 16: 30 p.

CARROLL, D. 1958. Role of clay minerals in the transportation of iron. Geochim. Cosmochim. Acta 14: 1–27.

CHAU, Y., V. K. CHAWLA, H. F. NICHOLSON, AND R. A. VOLLENWEIDER. 1970. Distribution of trace elements and chlorophyll *a* in Lake Ontario. Proc. Conf. Great Lakes Res. 13: 659–672.

CRONAN, D. S. AND R. L. THOMAS. 1972. Geochemistry of ferromanganese oxide concentrations and associated deposits in Lake Ontario. Geol. Soc. Am. Bull. 83: 1493–1502.

DAVIES, A. G. 1970. Iron, chelation and growth of marine phytoplankton. J. Mar. Biol. Assoc. U.K. 50: 65–86.

DELFINO, J. J., AND G. F. LEE. 1968. Chemistry of manganese in Lake Mendota, Wisconsin. Environ. Sci. Technol. 2: 1055–1148.

GARRELS, R. M., AND C. L. CHRIST. 1965. Solutions, minerals and equilibria. Harper and Row, Publishers. New York, N.Y. 400 p.

HUTCHINSON, G. E. 1957. A treatise on limnology. John Wiley & Sons, Inc. New York, N.Y. Vol. 1: 900.

KEMP, A. L., R. L. THOMAS, C. I. DELL, AND J-M. JAQUET. 1976. Cultural impact on the geochemistry of sediments in Lake Erie. J. Fish. Res. Board Can. 33: 440–462.

KUAUSKOPF, K. B. 1957. Separation of manganese from iron in sedimentary processes. Geochim. Cosmochim. Acta. 12: 61–84.

LAM, D. C. L., AND J-M. JAQUET. 1976. Computations of physical transport and regeneration of phosphorus in Lake Erie. J. Fish. Res. Board Can. 33: 550–563.

LEWIN, J., AND C. H. CHEN. 1973. Changes in the concentration of soluble and particulate iron in seawater enclosed in containers. Limnol. Oceanogr. 18: 590–596.

MCMAHON, J. W. 1969. The annual and diurnal variation in the vertical distribution of acid-soluble ferrous and total iron in a small dimitic lake. Limnol. Oceanogr. 14: 357–367.

MORTIMER, C. H. 1941. The exchange of dissolved substances between mud and water in lakes. J. Ecol. 29: 280–329.

1942. The exchange of dissolved substances between mud and water in lakes. J. Ecol. 30: 147–201.

MURPHY, T. M., AND D. R. S. LEAN. 1975. The distribution of iron in a closed ecosystem. Verh. Int. Ver. Limnol. 19: 258–266.

NRIAGU, J. O. 1974. Effects of Eh and pH on the dissolution of freshwater ferromanganese modules. Earth Research. 2: 25–34.

NRIAGU, J. O., AND C. I. DELL. 1974. Diagenetic formation of iron phosphates in recent lake sediments. Am. Mineral. 59: 934–946.

ROSSMAN, R., AND E. CALLENDER. 1968. Manganese nodules in Lake Michigan. Science. 162: 1123–1124.

STUMM, W., AND J. J. MORGAN. 1970. Aquatic chemistry. John Wiley & Sons, Inc. New York, N.Y. 583 p.

TRAVERSY, W. J. 1971. Methods for chemical analysis of waters and wastewaters. Inland Waters Br. Dep. Fish. For. Tech. Rep. 169 p.

Sediment Yields and Nutrient Loadings From Canadian Watersheds Tributary to Lake Erie: an Overview

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ONGLEY, E. D. 1976. Sediment yields and nutrient loadings from Canadian watersheds tributary to Lake Erie: an overview. J. Fish. Res. Board Can. 33: 471–484.

Mean annual loadings and unit yields for sediment, phosphorus, nitrogen, and chloride have been calculated for 19 river-mouth locations on the Canadian side of Lake Erie for the period 1967–72. Data, drawn from routine monitoring data files of Federal and Provincial agencies, are discussed in terms of biases inherent in such monitoring programs. Unit yields are only marginally related to basin substrate texture but show pronounced storage characteristics within basins for sediment and sediment-bound nutrients. It is suggested that there is a minimum threshold size of basin below which nutrient runoff is relatively unaffected by basin storage and uptake.

ONGLEY, E. D. 1976. Sediment yield and nutrient loadings from Canadian watersheds tributary to Lake Erie: an overview. J. Fish. Res. Board Can. 33: 471–484.

Nous avons calculé les charges et les rendements à l'unité moyens annuels pour le sédiment, le phosphore, l'azote et le chlorure à 19 embouchures de rivières sur le côté canadien du lac Erié au cours de la période 1967–72. Nous analysons les données cueillies de façon routinière et continue, et extraites des filières d'organismes fédéraux et provinciaux, en termes d'erreurs systématiques inhérentes à de tels programmes de surveillance continue. Les rendements à l'unité n'ont qu'une relation marginale avec la texture du substrat des bassins. Dans les limites d'un bassin, cependant, ils montrent des caractéristiques prononcées de stockage de sédiment et de substances nutritives fixées au sédiment. Il y aurait, croyons-nous, une grandeur de seuil minimale pour un bassin au-dessous de laquelle le ruissellement des nutritifs serait relativement indépendant du stockage et de l'assimilation par le bassin.

Accepted November 17, 1975 Accepté le 17 novembre 1975

BECAUSE of its central importance to water quality in the Lower Great Lakes, Lake Erie has probably received more scientific attention than most bodies of water of similar size. Apart from massive limnological research and survey studies of input loadings (Environmental Protection Agency 1971; Great Lakes Water Quality Board 1973; International Joint Commission 1969), numerous studies (some of which are referenced below) have been directed towards water quality within individual tributary basins. Recognizing the importance of land drainage as an essential part of any explanation and therefore control of pollution in the Great Lakes, an International Reference Group on land drainage was established in 1972 (Pollution from Land Use Activities Reference Group — PLUARG) following the signing of the Great Lakes Water Quality Agreement in that same year.

Aspects of the PLUARG (Canadian) program which are pertinent to this study are a compilation of land use categories across agricultural southern Ontario, detailed watershed studies to document the interrelationships between specific land uses and water quality, and an evaluation and compilation of loadings from all available data pertaining to (Canadian) sediment and nutrient concentrations as routinely monitored at river mouths. Utilizing relationships found in detailed watershed studies, it is anticipated that impact of land use on Great Lakes water quality can be calculated or at least inferred for the balance of southern Ontario. Although the first and third aspects of the program are nearing completion, the second (detailed watershed studies) is still in progress; hence, the detailed information required to assess the adequacy of the routine monitoring data used herein is not yet in hand. Therefore, the overview presented in this report is not definitive; rather, it represents a 'first look' at regional water quality data files. Despite limitations (discussed below) of available data files, spatial and temporal trends which emerge not

Printed in Canada (3999)
Imprimé au Canada (J3999)

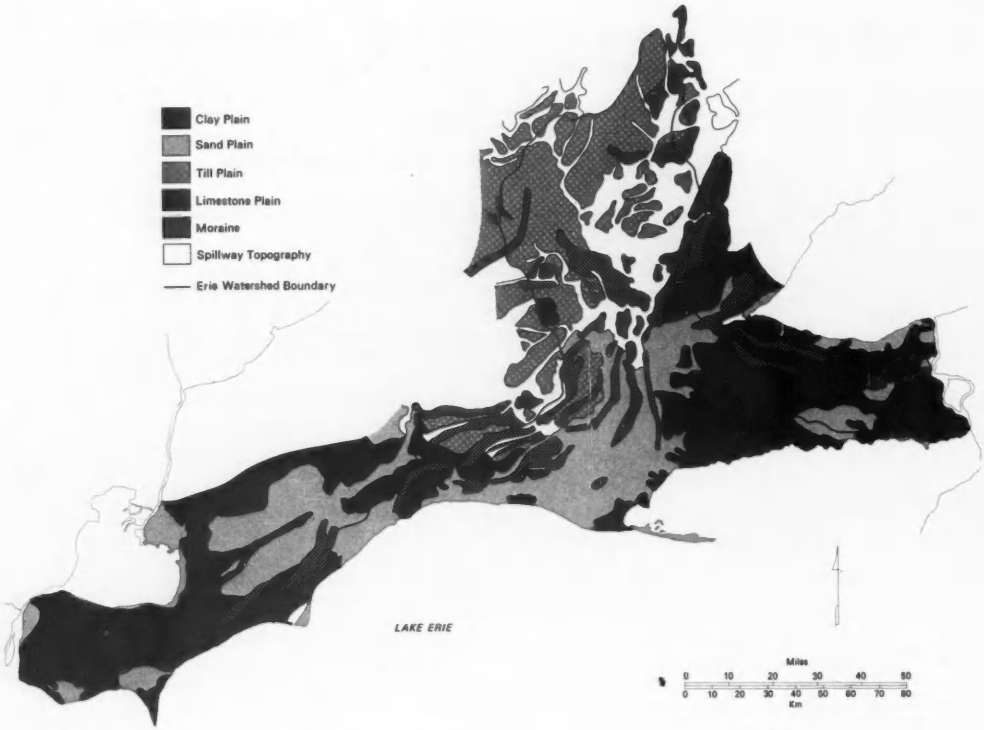

FIG. 1. Physiography of the (Canadian) Lake Erie watershed (after *Physiography of Southern Ontario*, Department of Mines and Technical Surveys, 1:1,000,000, 1965).

only provide the first regional description of tributary loadings but also draw attention to relationships which merit closer examination.

Description of Study Area

The structural and stratigraphic composition of the Erie basin, the Pleistocene physiographic and sedimentary records, and the post-colonial cultural development within the Erie watershed have been summarized by Sly (1976). Because of the hydrologic differences associated with clay versus sand substrates, it is useful for the purpose of this overview of land drainage to draw attention to the existence of widespread lacustrine clay plains to the west and east of the Erie watershed, and the extensive deltaic sands occupying the central sector (Fig. 1). As noted by Sly (1976), most of the (Canadian) Erie watershed is rural, representing diffuse sources of nutrient and sediment loads. These loads may, however, reflect point sources in several basins (Fig. 2) which contain urban development to a greater or lesser degree (#58 Grand River; #62 Lynn River: Simcoe; #69 Big Otter Creek: Tilsonburg; #71 Catfish Creek: Aylmer; #72 Kettle Creek: St. Thomas). Also commented upon by Sly (1976) and reflected in Fig. 2 is the paucity of well-developed drainage, particularly on the clay plains. Not only is the drainage texture coarse in monitored basins, but is generally ill-defined in the interbasin areas reflecting low gradients and immature development of drainage in post-Pleistocene time.

Data Base

Large data files consisting of discharge and concentration values of particulate and soluble materials were made available, respectively, by the Water Survey of Canada and the Ontario Ministry of the Environment. These files represent, in part, an ongoing program to determine hydrologic and nutrient mass balances in the Great Lakes. The data base represents most natural Canadian land drainage to Lake Erie and encompasses some 23

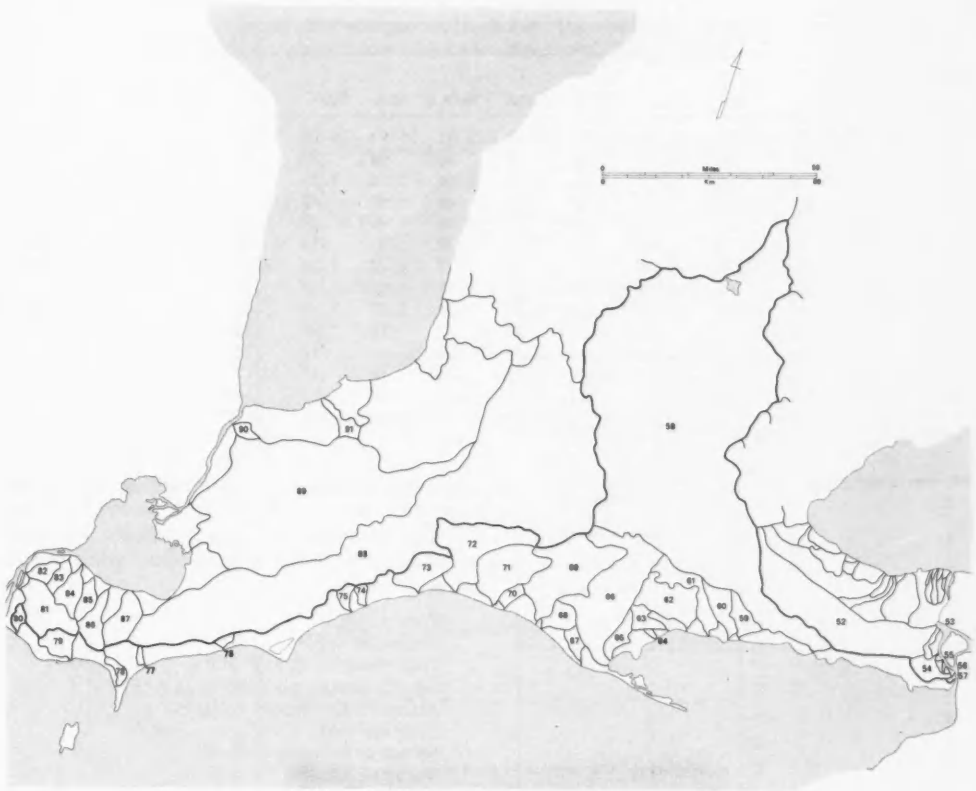

FIG. 2. Index map of monitored Canadian basins tributary to the Great Lakes (Lake Erie Region). Basin numbers are referred to in the text.

basins (Fig. 2) ranging in size from 10 to 6653 km². The Detroit River is excluded from these data. Nineteen of the basins are monitored for water quality and 13 have continuous discharge records as noted below:

a bc			
58:11	Grand River	69:11	Big Otter Ck.
59:01	Stoney Ck.	70:10	Silver Ck.
60:01	Sandusk Ck.	71:11	Catfish Ck.
61:11	Nanticoke Ck.	72:11	Kettle Ck.
62:11	Lynn River	73:01	Talbot Ck.
63:10	Young Ck.	74:01	Brock Ck.
64:10	Fishers Ck.	75:01	16 Mile Ck.
65:11	Dedrich Ck.	76:10	OAC nr. Merlin*
66:11	Big Ck. (Norfolk)	77:01	Muddy Ck.
67:01	Clear Ck.	78:11	Sturgeon Ck.
68:11	South Otter Ck.	79:01	Cedar Ck.
		80:01	Big Ck. (Essex)

[a]Basin number; [b]Federal Gauge Information: 0 = No gauge, 1 = Gauge; [c]Water Quality: 0 = No data, 1 = Data; *discontinuous record.

Discharge records vary in length between 2 yr (#78 Sturgeon Creek) and 47 yr (#58 Grand River) for the period through 1972. Mean monthly discharge varies between 0.06 and 175 m³/s, and mean annual discharge from 0.21 to 67 m³/s (Table 1). Canadian tributary drainage to Lake Erie is dominated by the Grand River, Ont. Of the total mean annual discharge to Lake Erie the Grand accounts for 68%. Although descriptive statistics have not been calculated, monthly mean[1] flow regimes are highly variable from one year to another. Illustrative of this variability are the 1970 flow data for the Grand River which, although the flow is highly

[1]The terms 'monthly mean' and 'mean monthly' should not be confused. The former describes an average condition within a single month whereas the latter is the average of a number of specific months. The terms 'annual mean' and 'mean annual' have parallel meaning.

TABLE 1. Mean monthly and annual discharge by basin (m^3/s).

Basin no.[a]	Jan.	Feb.	Mar.	Apr.	May	June	July	Aug.	Sept.	Oct.	Nov.	Dec.	Year
58	57.15	63.57	167.92	175.65	66.65	39.18	33.92	26.89	26.90	33.89	47.17	64.42	66.89
61	.87	1.46	4.96	3.55	1.52	.74	.40	.24	.20	.46	1.28	2.21	1.44
62	2.82	3.23	5.25	4.77	3.21	2.29	2.01	1.76	1.59	1.66	2.16	2.70	2.74
63	.81	.87	1.29	1.15	.75	.53	.44	.45	.45	.50	.63	.78	.73
64	.21	.22	.28	.23	.20	.19	.18	.19	.20	.22	.21	.22	.21
65	1.13	1.30	2.07	1.85	.99	.51	.33	.39	.24	.34	.85	1.21	.95
66	6.60	8.04	14.20	13.70	8.56	5.11	3.77	3.72	3.78	3.92	5.32	7.63	7.02
68	1.59	1.85	3.01	2.31	1.23	.82	.51	.46	.42	.58	1.16	1.70	1.31
69	9.40	11.70	16.85	13.87	6.86	4.31	3.19	2.87	2.70	3.46	4.96	8.20	7.45
70	.33	.54	1.22	.71	.38	.26	.22	.18	.20	.24	.41	.52	.43
71	5.25	7.45	11.91	9.06	2.68	2.31	.98	.93	.37	.75	3.08	6.00	4.23
72	5.52	8.02	10.30	8.56	4.13	4.77	.74	.74	.52	2.17	4.04	6.04	4.34
78	.22	.16	.67	.59	.20	.06	.08	.07	.07	.16	.43	.50	.35

[a]No data for basins 59–60, 67, 73–77, and 79–80.

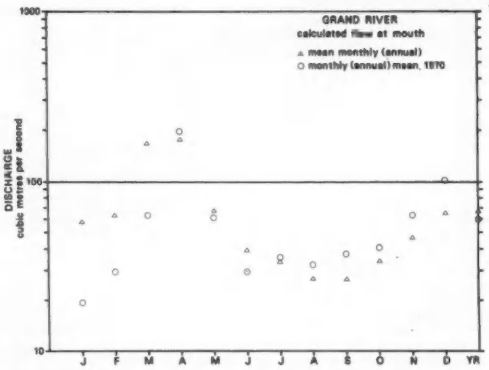

FIG. 3. Discharge record for mouth of Grand River.

controlled, demonstrate up to a threefold difference between mean monthly and the 1970 monthly mean flow (Fig. 3). The 1970 data were not selected to display any particular extremes; rather, that year is presented merely for compatibility with other information presented by Burns (1976). Seasonality of flow, illustrated by the Grand River (Fig. 3) reflects a dominant spring melt with greatly reduced flows in summer. The mean monthly data of Table 1 give an inadequate impression of hydrologic processes, in that episodic melt periods during the entire winter serve to carry off much of the winter precipitation. On smaller creeks, episodic melts produce multiple ice layers with separated interlayer flow paths. In addition, such values also camouflage the importance of short-duration, but intense, summer storm phenomena which are responsible for flushing much of the nonmelt suspended sediment and, probably, a considerable quantity of particulate-bound nutrient into Lake Erie.

Records of concentrations of solids and soluble materials were initiated in or after 1964 (Table 2). The concentration data routinely available at each monitored site and included in this report are as follows:

Total solids
Suspended solids
Total phosphorus as P (TP)
Soluble reactive phosphorus as P (SRP)[2]
Ammonia as nitrogen (NH_3-N)
Total kjeldahl[3]
Nitrate as nitrogen (NO_3-N)
Nitrite as nitrogen (NO_2-N)
Chloride

Dissolved solids may be obtained by subtracting suspended from total solids.

Occasionally available are:

Hardness as $CaCO_3$
Total iron
Alkalinity as $CaCO_3$

Those water quality data incorporated into this report are generally taken at or near the mouth of each creek or river (#75 Sixteen Mile Creek is an exception). Although all parameters are not necessarily measured on each sample date the numbers of samples per month and in total for total solids is indicative of numbers of samples for the other eight parameters routinely available (Table 2). With the exception of the Grand River, water quality data are obtained from grab samples taken from less than a meter depth. The Grand River data have been averaged from four sample values taken, respectively, from the left and right banks, and at the surface and at depth.

[2]Values should be read with caution as field samples were bottled and sent to the laboratory for analysis.

[3]Equivalent to total organic nitrogen + (NH_3-N).

TABLE 2. Water quality sample record by basin for period of record.

No.	Basin name[a]	Period of record	J	F	M	A	M	J	J	A	S	O	N	D	Total	Year mean
58	Grand River	1964–71	5	5	5	5	5	7	7	7	6	6	5	6	69	8.6
59	Stoney Ck.	1964–72	8	6	8	9	8	8	8	4	4	6	8	6	83	9.2
60	Sandusk Ck.	1964–72	8	6	8	9	9	8	10	8	6	7	9	6	94	10.4
61	Nanticoke Ck.	1964–72	8	6	8	9	9	8	10	8	6	8	9	6	95	10.6
62	Lynn River	1964–72	8	6	8	9	9	8	9	9	7	8	9	6	96	10.7
65	Dedrich Ck.	1964–72	8	5	7	9	9	8	10	9	7	8	9	6	95	10.6
66	Big Ck. (Norfolk)	1964–71	6	5	7	7	7	7	9	8	6	8	7	6	83	10.4
67	Clear Ck.	1964–72	7	5	8	9	9	8	10	9	7	7	9	6	94	10.4
68	South Otter Ck.	1964–71	6	6	7	7	7	7	9	8	6	7	7	6	83	10.4
69	Big Otter Ck.	1964–71	6	5	7	8	7	7	9	8	6	8	7	6	84	10.5
71	Catfish Ck.	1964–71	6	6	6	8	6	7	8	9	6	8	6	7	83	10.4
72	Kettle Ck.	1964–72	7	7	8	8	8	9	8	12	7	9	6	8	97	10.8
73	Talbot Ck.	1964–71	4	6	7	8	7	10	7	8	5	8	6	5	81	10.1
74	Brock Ck.	1968–72	4	4	6	7	5	6	5	6	4	6	5	3	61	12.2
75	16 Mile Ck.	1964–72	6	7	8	10	7	11	7	10	7	8	8	5	94	10.4
77	Muddy Ck.	1966–71	3	6	7	9	6	9	6	8	5	6	9	4	78	13.0
78	Sturgeon Ck.	1966–72	4	6	7	9	6	10	6	7	6	8	9	4	82	11.7
79	Cedar Ck.	1964–71	6	6	7	9	6	10	7	9	6	6	6	5	83	10.4
80	Big Ck. (Essex)	1964–71	6	7	7	9	6	11	6	9	6	6	7	5	85	10.6

[a]No data for basins 63, Young Ck.; 64, Fishers Ck.; 70, Silver Ck.; 76, OAC nr. Merlin.

Loadings (Table 4) are reported herein in metric tons (1000 kg) and are calculated in one of two ways depending upon the availability of discharge records.

1) *Basins with quality and discharge records* — The reported mean annual loadings are derived from individual monthly loading values over the period of quality record. Monthly loadings (Equation 1) in metric tons (L_m) are derived from monthly mean concentrations in grams per liter (C_m), monthly mean flow (Q_m), a constant ($k = 2.4$) and numbers of days per month (n).

$$L_m = C_m Q_m k\ n \quad (1)$$

2) *Basins with quality but no discharge records* — The reported mean annual loadings (Equation 2) are obtained from annual loadings (L_a) which, in turn, are derived from annual mean concentration (C_a), mean annual flow (Q_a, as described below), a constant (k) and number of days per year (n).

$$L_a = C_a Q_a k\ n \quad (2)$$

Limitations of the Data Set

Bearing in mind the nature of the data files, a number of limitations must be stated at the outset. Subject to these limitations, the results are thought to indicate trends but could be seriously in error with regard to absolute values.

1) Water quality data were collected by the grab sample technique. This technique, while it does not seriously affect solute determinations, seriously underestimates suspended load which is depth dependent. It is not known whether the adsorbed portion of certain nutrient loadings (TP, for example) is seriously underestimated for an unresolved question is the degree to which aggregates of nutrient-bearing fine-grained particulates are depth dependent under different flow regimes.

2) High flow events, particularly the short summer storm phenomena, are frequently not represented in Ministry concentration data. Two ramifications emerge; one relates to the overestimation of solute loads through the use of low flow and therefore higher than average concentration data. Such an overestimation has recently been demonstrated by Neuhold (1975) over a 10-wk summer period in Highland Creek. Pending completion of detailed basin studies in the Erie watershed, it is not possible to establish relative accuracy of solute data used herein.

The second ramification bears upon the magnitude-frequency concept of erosional processes. Bias towards low flow data does not permit evaluation of infrequent moderate to high flow events which are responsible for the bulk of suspended sediment transport. Therefore, serious underestimation of clastic loads is inevitable. Illustrative of clastic underrepresentation are data provided by the sediment survey program of Canada (Stichling 1974). Using temporally distributed data, mean concentration and mean yield for Big Otter Creek (#69) over the 5-yr period of record are, respectively, mapped in the ranges of 210–400 ppm and 88–350 metric tons·km^{-2}·yr^{-1}. Comparable values using Ministry data herein amount to 55 ppm and 22 metric tons·km^{-2}·yr^{-1}. Of the seventeen Federal sediment survey stations within Ontario, the only other useable station within the Erie watershed is that within Big Creek basin (#66). Values herein are only slightly more representative than for Big Otter Creek — 30 ppm and 11 metric tons·km^{-2}·yr^{-1} versus 51–200 ppm and 18–88 metric tons·km^{-2}·yr^{-1} found by the sediment survey program. In either case, yield estimates obtained from Ministry data are in the range of one-half to one-tenth of the more accurate information from the sediment survey program.[4] Dickenson (personal communication) has found no direct or consistent relationship between Ministry and Federal sediment data sets. Bearing in mind the differences in provenance and transport mechanisms, the solute data are thought to be less biased than suspended load.

Associated with underrepresentation of clastic load is the fact that significant quantities of nutrient are adsorbed on fine-grained particulates. Hence, certain aspects of nutrient loadings may well also be seriously underestimated. In a recent study of phosphorus transport dynamics in the Brandywine basin of Pennsylvania and Delaware, Cahill et al. (1974) found that more than half the TP load was associated with storm discharge. During such events particulate and organic-bound inorganic phosphorus are mobilized within the watershed, greatly decreasing the soluble – TP ratio. If these relationships are found to hold for Lake Erie watersheds, Ministry data may be in error by 50% for TP and by something less for soluble phosphorus.

3) The calculation of loadings requires discharge and concentration data. However, discharge and quality records are seldom obtained for the same site. Whereas water quality samples are usually taken close to the stream mouth, the

[4]The comparative values are only indicative of magnitudes of error. There is no direct relationship between Ministry suspended sediment and Federal sediment survey data for the former do not discriminate between organic and mineral sediment whereas the latter refer only to mineral sediment.

TABLE 3. Coefficients of determination for mean monthly and mean annual discharge (Y)[a] against basin area above gauge (X)[b]. (Source: Ongley 1974.)

Lake	N[c]	Jan.	Feb.	Mar.	Apr.	May	June	July	Aug.	Sept.	Oct.	Nov.	Dec.	Annual
Ontario	23	93.1	92.3	94.6	95.1	94.2	86.9	79.3	66.2	71.4	87.9	93.3	94.7	96.5
Erie	9	96.7	96.1	98.9	99.2	98.8	96.7	88.3	84.0	81.6	93.9	99.2	97.9	99.6
Huron	19	84.5	92.6	85.7	93.4	91.0	88.5	86.9	91.8	83.4	94.3	93.3	92.8	97.5
Superior	7	92.5	91.6	90.0	77.2	77.6	88.6	97.3	96.6	95.7	94.0	92.7	92.8	82.5

[a] Log_{10} discharge.
[b] Log_{10} area.
[c] N = number of gauge records used.

discharge recorder may be located some distance upstream to avoid backwater effects. As noted below, discharge is a simple function of area, but water concentration data may vary widely over short distances, reflecting diffuse and point sources. Although it is a relatively straightforward matter to adjust for flow, any attempt to predict systematic spatial changes in concentration values would indeed be foolhardy.

To compensate for the natural increase in discharge between the gauge and water quality monitoring sites, I used the following justification and procedure in an earlier report (Ongley 1974) dealing with the four Canadian Great Lakes.

> Computation of yields and loadings at the mouth of each tributary basin should take into account the natural increase in flow between the gauge station and the stream terminus. Allowing that discharge is composed of a variety of components which are spatially and temporally controlled by atmospheric, topographic and geologic factors, it can be shown that discharge is statistically related to basin area at a monthly and annual level of aggregation. Coefficients of determination for power regressions of mean monthly and mean annual discharge in cubic feet per second against basin area upstream from the gauge (square miles) by Great Lake for gauge records closest to the tributary mouth, are large throughout the year [Table 3 in the present report] with the exception of late summer months in the Lake Ontario basin. All regression equations are highly significant. Area may be regarded as a composite variable incorporating and averaging out the individual effects of topography, geology and climatic variability.
>
> To better assess total discharge into the Great Lakes, those discharge stations located sufficiently upstream so that there is more than a ten percent difference in basin area as recorded at the water quality sites in comparison with area upstream of the discharge sites, are augmented by pro-rating monthly and annual flow values on a square mile basis.

Monthly (annual) mean values are employed in the absence of data permitting establishment of concentration–discharge relationships.

4) Many small basins which are tributary to the Great Lakes and for which there exists water quality information, lack discharge records. In view of the large coefficient of determination for the mean annual area–discharge relationship for all Great Lakes and for Lake Erie drainage in particular (99.6% — Table 3), it was decided to estimate annual flow for ungauged basins and to calculate annual loadings using annual mean concentration data. Following a rigorous statistical comparison of mean annual versus annual mean discharge records (Ongley 1974) it was established that there is no statistical validity in using estimated annual mean discharge. Therefore, annual loadings for ungauged basins are obtained from annual mean concentrations and mean annual discharge (Q_m) obtained from Equation 3 where A is Basin Area.

$$\text{Log}_{10}\, Q_m = 0.111 + 0.941\ \text{Log}_{10} A \qquad (3)$$
$$r^2 = 97.2\%$$

5) One of the most dubious assumptions employed herein is that monthly (annual) mean values for concentration data can be utilized to estimate monthly (annual) mean loadings. The writer is fully cognizant of the fact that the methods used produce values which bear no necessary regular relationship to actual values. Moreover, the use of mean values implies a known (and linear) mathematical relationship between concentration and discharge data; indeed, these relationships are not known for these data and are, in general, seldom linear. The justification for employing these methods lies in the fact that the paucity of data per tributary stream (less than two samples per month — Table 2) and the systematic sampling biases noted above preclude a more precise analysis. Additional data necessary to correct these biases are unlikely to be forthcoming for as wide an area as populated southern Ontario. Hence, one must adopt the pragmatic attitude that values calculated herein are best estimates based upon all available data to date. As part of an ongoing monitoring program, additional data from selected tributaries will, in

TABLE 4. Loadings: Lake Erie tributary drainage[a] as percent of total for the Lake in metric tons per km². (Source: Ongley 1974.)

Basin no.	Area (km²)	Total solids		Suspended solids		Dissolved solids		TP		SRP		NH_3-N		Total Kjeldahl		NO_2-N		NO_3-N		Chloride	
		%	Tons/km²	%	Tons/km²	%	Tons/km²	%	Tons/km²	%	Tons/km²	%	Tons/km²	%	Tons/km²	%	Tons/km²	%	Tons/km²	%	Tons/km²
58	6653	65.8	134.3	53.9	12.5	67.3	121.7	87.1	.203	91.7	.149	74.7	.098	69.4	.367	64.1	.010	60.0	.35	68.7	8.8
59	117	1.6	181.6	0.7	9.5	1.7	172.1	0.5	.066	0.4	.035	0.9	.063	1.4	.413	1.1	.010	1.0	.32	3.4	25.0
60	160	1.6	136.2	2.0	19.7	1.6	116.5	0.6	.056	0.3	.022	1.2	.066	1.8	.402	1.5	.009	1.5	.36	2.0	10.5
61	183	1.3	98.0	1.0	8.1	1.4	89.7	0.3	.024	0.2	.009	0.7	.031	1.2	.234	1.0	.006	1.5	.31	1.3	5.9
62	287	2.6	120.8	1.7	8.8	2.7	111.9	1.1	.059	0.9	.035	2.9	.087	2.4	.294	3.4	.012	2.4	.32	2.2	6.4
63																					
64																					
65	81	0.7	108.4	0.7	13.0	0.7	95.4	0.1	.024	0.1	.006	0.3	.031	0.5	.206	0.3	.004	0.4	.20	0.3	2.6
66	730	5.7	105.7	5.0	10.6	5.8	95.0	1.0	.021	0.4	.005	2.4	.028	3.5	.168	3.2	.005	4.5	.24	3.8	4.4
67	51	0.5	124.0	0.6	17.5	0.5	106.4	0.1	.021	0.0	.005	0.2	.028	0.3	.227	0.2	.005	0.4	.27	0.2	2.4
68	99	1.0	130.9	1.3	19.8	0.9	111.0	0.6	.098	0.1	.005	0.3	.028	1.0	.343	0.4	.004	0.6	.23	0.3	2.9
69	705	6.4	123.6	9.9	21.7	6.0	101.8	1.6	.035	0.8	.012	3.1	.038	4.7	.234	4.5	.006	9.0	.50	4.6	5.5
70																					
71	387	4.7	163.4	10.8	43.1	3.9	120.2	1.5	.059	1.1	.029	2.6	.059	3.1	.273	4.4	.011	6.5	.66	3.0	6.6
72	441	3.5	108.0	3.8	13.2	3.5	94.7	2.6	.091	2.4	.059	4.6	.094	4.2	.332	7.9	.018	6.2	.54	5.4	10.5
73	151	1.9	167.5	4.5	45.5	1.5	121.9	0.7	.066	0.2	.017	1.7	.101	2.0	.451	2.7	.018	2.9	.75	1.3	7.5
74	23	0.3	177.7	0.3	22.1	0.3	155.5	0.4	.287	0.4	.199	0.5	.189	0.4	.567	1.0	.045	0.4	.67	0.4	13.6
75	37	0.5	188.6	0.4	18.4	0.5	170.2	0.6	.231	0.4	.122	1.0	.234	0.7	.682	1.1	.031	0.5	.48	0.5	11.8
76																					
77	10	0.1	158.1	0.3	41.5	0.1	116.6	0.3	.525	0.2	.176	1.1	.913	0.7	2.359	0.3	.029	0.1	.43	0.1	11.9
78	32	0.5	190.5	0.7	35.0	0.4	155.4	0.3	.140	0.2	.063	0.3	.087	0.4	.392	1.3	.040	1.0	1.23	0.6	15.6
79	103	0.8	108.5	1.9	27.6	0.7	80.8	0.4	.056	0.2	.015	0.9	.077	1.2	.395	1.1	.010	0.9	.32	1.1	8.7
80	51	0.7	189.7	0.6	18.9	0.7	170.8	0.3	.094	0.1	.017	0.7	.122	1.4	.959	.05	.011	0.4	.27	1.1	16.9
Total for Lake	10301	100.0	131.8[b]	100.0	15.0[b]	100.0	116.7[b]	100.0	.15[b]	100.0	.11[b]	100.0	.09[b]	100.0	.34[b]	100.0	.01[b]	100.0	.38[b]	100.0	8.27[b]
Total metric tons			1359.9		155.4		1204.4		1.6		1.1		.9		3.5		.1		3.9		86.3

[a]Detroit River is excluded.
[b]Weighted by tributary area.

future, indicate the degree to which the present calculations are in error. Accepting the fact that absolute values are inevitably in error one is tempted to infer relative status for the calculated loads. Although Ministry concentration data are generally much less variable (per parameter) between streams than discharge, loads (as products of concentration and discharge) may reflect relative differences in discharge characteristics. This is probably a fair assumption for the end members of the data set but is inadequate for basins of similar area (and therefore discharge).

With no expectation of improved regional water quality records, I chose to explore a limited number of relationships amongst calculated loadings, basin area, and basin substrate. In view of data inadequacies these relationships are admittedly speculative but, subsequent to current detailed studies of land use and water quality in limited areas of southern Ontario, they may suggest either the degree to which routinely monitored data may be used to discriminate amongst gross regional differences or the fallibility of employing such data. In any case, one might take some solace in the knowledge that the land drainage directly into the Canadian side of Lake Erie is much less significant in terms of deleterious effects than the contribution of the Detroit River (Great Lakes Water Quality Board 1973).

In this report, two points of view are taken.

1) Great Lakes water quality is a function of input loadings, that is, the total annual tonnage of any particular sediment or nutrient parameter discharged into the nearshore zone. Hence, the estimated total annual loadings of the 10 parameters routinely available are recorded in Table 4. Individual contributions by tributary are recorded as percentages of the total.

2) From a land-based point of view, absolute values of loadings are of little significance without reference to the size of the basins. Total tonnage per parameter, pro-rated on a unit area basis, provides a spatially averaged measure of intensity of parameter production. Although such values are, in reality, highly influenced by point sources, pro-rated values at least allow a measure of interbasin comparison where the paucity of point source information precludes a deterministic evaluation of concentration data[5] (Table 4).

Data Presentation

For the purpose of illustrating trends, data are presented in the form of scatter diagrams in which parameter yield in metric tons per square kilometer (unit yield) is plotted against basin area. Because of data uncertainty, the plots have not been subjected to regression or cluster analyses. The scatter diagram has several advantages; the unit yield permits an interbasin evaluation and yet, basins of similar size and, therefore, discharge are easily grouped together for comparison. In the latter instance the unit area tonnage is an indication of relative total loadings.

Most of the tributary basins in the Lake Erie watershed can be divided into predominantly coarse (sand) or fine-grained (clay) overburden,[6] thus suggesting gross permeability differences and, therefore, differing potentials for production of overland flow versus interflow (throughflow). Although these data do not generally partition into distinct groupings, textural differentiation by basin may serve as a surrogate for hydrologic response, and differing crop types and land management practices.

Solids

Total Solids

The 1.36 million tons of total solids delivered annually to Lake Erie from Canadian tributary drainage is at least 21% larger than that estimated by the Great Lakes Water Quality Board (1973), but is only about 2% of total solids input from the Detroit River. The Grand River accounts for 65.8% of the 1.36 million tons.

Suspended Solids

Although suspended sediment is only a fraction of the total sediment yield of Great Lakes tributary rivers, the presence of significant quantities of particulate material is a recent phenomenon associated with the post-colonial period. Studies such as those of Kemp et al. (1974) have demonstrated a several-fold increase in recent sedimentation rates into the Great Lakes. The real increase in erosion is likely to be even higher than lake sedimentation rates indicate, as coarse-grained materials are likely to remain in the nearshore zone because of loss of stream competence at the lake boundary. Detailed data on erosion and sediment yields within the Great Lakes basin are meager. Nevertheless, estimates of suspended sediment yield or denudational equivalent have been made by the International Joint Commission (PLUARG — Task A, 1974) for each of the Great Lakes (U.S. side) and by

[5]Concentration data by month, year, and period of record are recorded in Ongley (1974).

[6]Land Classification maps, Ontario Ministry of Natural Resources (scale 1:250,000).

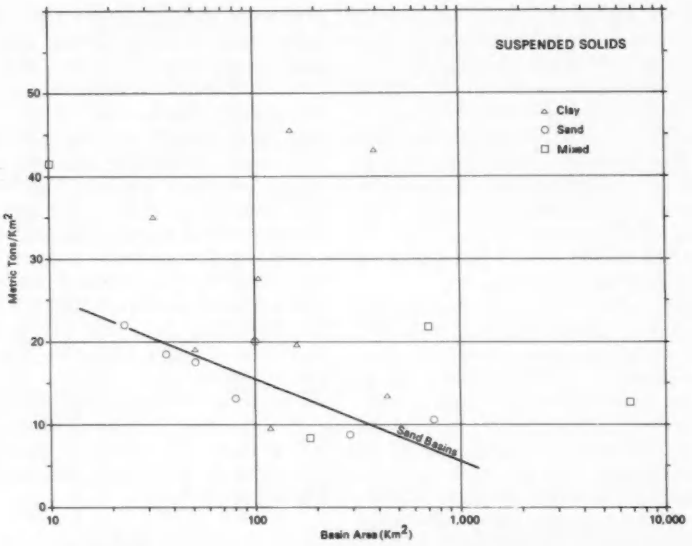

FIG. 4. Mean annual unit yield of suspended solids against basin area.

Ongley (1973) for Lake Ontario. Other estimates from much more generalized information have been made by Fournier (1960), Slaymaker and McPherson (1973), and Stichling (1973, 1974). From the point of view of productivity and eutrophication of the Great Lakes, suspended sediment is of interest primarily for its ability to transport significant quantities of adsorbed pesticides, plant nutrients, nuclear fallout, and infectious particles (PLUARG — Task A, 1974, p. 11).

For reasons noted above, the 155,000 tons of suspended solids may be from one-half to one-tenth of actual value. Nevertheless, the scatter plot (Fig. 4) illustrates two concepts worthy of notice. First, the inverse relationship between unit yield and basin area serves to demonstrate the storage function within basins. Positively related to basin size, storage is, in part, dependent upon topographic characteristics and composition of the fluvial system. Stored sediment may be removed entirely from fluvial transport or partially remobilized during periods of high discharge. Inevitably, the net effect is to reduce unit yields in proportion to basin size (Brune 1968; Ongley 1973). Second, in the absence of urban sources one might reasonably expect in the Lake Erie watershed that, with comparable flow, a river system draining fine-grained sediments will carry a greater suspended load than one draining coarse deposits both because of the greater tendency for surface flow on clay, and, therefore, mobilization of sediment, and the increased capacity of a stream with decreasing particle size. Although the present data do not distinguish between organic mineral sediment making the trends illustrated in Fig. 4 difficult to interpret, the basins separate into clay and sand groups. The latter have a lower yield and a negative relationship with basin area.

DISSOLVED SOLIDS

Expectably, at least for these data, no trends relating unit yield to solute load to basin area or substrate emerge. Nutrient loadings, being of primary interest within the solute component, are neither simply related to discharge nor land use within this data set. The Grand River accounts for 67.3% of the mean annual 1,328,000 tons of dissolved solids delivered annually to Lake Erie.

Nutrient Loadings

As nitrogen and/or phosphorus are generally considered to be the principal growth controlling nutrients, the following discussion applies to both. The relationship between stream-mouth loadings of N and P species and provenance is complex. Although progress has been made in the reduction of phosphorus, reduction in nitrogen loadings awaits a better understanding of nitrogen flow paths both within soil and in open channel flow. Indeed, in summarizing the literature to 1967, Hem (1970) reports that river-water concentra-

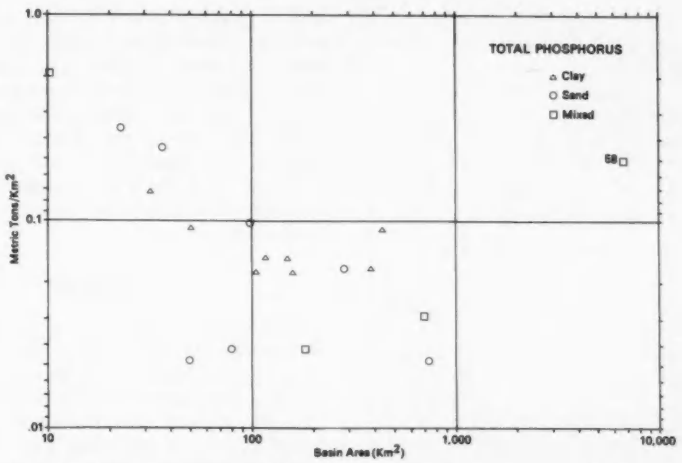

FIG. 5. Mean annual unit yield of total phosphorus (TP) against basin area.

tion data do not illustrate an increase commensurate with increased use of nitrogen fertilizers. Causality inferred from empirical relationships of loadings with land use categories upstream may therefore be misleading.

In their recent state-of-the-art assessment of nutrients from agricultural activities, Armstrong et al. (PLUARG — Task A, 1974) consider that crop cover may be much less important than farming practices and hydrologic factors. Studies by Johnson et al. (1965) and Carter et al. (1971) indicate that most nitrogen loss is through soil seepage, whereas more than half the phosphorus loss was monitored in surface runoff. This suggests that soil permeability (texture) may correlate positively with total nitrogen and negatively with phosphorus alone in plot or small basin studies. Similarly, TP loadings which include clay-bound phosphorus should be expected to increase in proportion with overland flow as soil texture becomes finer. Whether such simplistic relationships can be meaningfully established for whole basins for annual periods is not certain when nutrient transfer is also dependent upon characteristics of sink areas, uptake, episodic mobilization of organic and sediment-bound nutrient, temporal variations in nutrient availability, both positive and negative relationships with discharge for differing nutrient components, and the uncertain question as to basin area which contributes to storm runoff. Indeed, in their recent study of Brandywine Creek, Cahill et al. (1974) could only predict phosphate mass balance on a daily basis to within 25% of the actual value when all point sources were identified.

In their 1973 annual report, the Great Lakes Water Quality Board estimate that Canadian tributary drainage accounts for 12.8% of Canadian and American TP input into Lake Erie (5.5% if Detroit River is included). However, that report does not distinguish between contributions of municipal point sources and agricultural diffuse sources to tributary loadings of TP. Incorporating data from Weibel (1969), Armstrong et al. (PLUARG — Task A, 1974) calculate that only 15% of the TP input into Lake Erie (presumably for the US side) is produced by agricultural or rural runoff. In comparison, 77% originates from municipal or industrial wastewater.

In light of the findings of Armstrong et al. (PLUARG — Task A, 1974), one might expect to find large values of TP associated with the urban development in the Grand and several other basins and a second group of much lower values associated with diffuse rural sources. With the exception of the Grand River (#58) where two-fifths of river mouth values are estimated by MacCrimmon and Johnson (1971) to be attributable to the Electric Reduction Company of Canada Ltd, TP shows no such differentiation (Fig. 5). This observation is not only consistent with findings of Terry and Salbach (1969) who conclude that urban impact within the Catfish Creek catchment is minimal but also lends support to Sly's (1976) suggestion that diffuse agricultural sources may be considerably underestimated. Not surprisingly, TP is negatively related to basin area illustrating the partial dependence of TP values on particulate transfer. TP is unrelated to basin texture at this level of generalization.

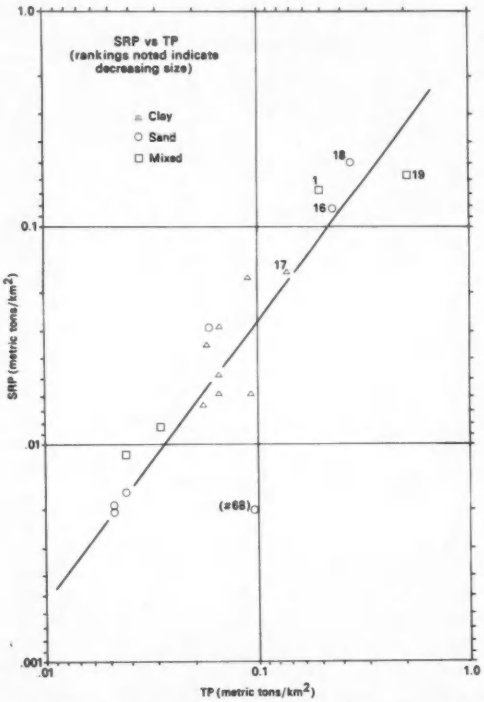

FIG. 6. Mean annual unit yields of soluble reactive phosphorus (SRP) as a function of total phosphorus (TP) yields.

A plot of SRP as a function of TP (Fig. 6) displays remarkable linearity. The best-fit line indicates that the ratio of SRP to TP increases with increasing TP. Whether this is unique for these biased data or holds true for more representative data sets is not known. Although textural groupings are not immediately apparent on the plot, the four smallest basins (#16–19), together with the Grand River (#1), represent the upper range of values. As noted for TP, the Grand River is an exception within this data set owing to significant point sources of phosphorus along its length and near its mouth. If one associates these upper values with a subset of basins denoted by a minimum basin size below which the usual relationships amongst size, runoff behavior, basin storage, and overburden texture do not hold, there remains a grouping of sand basins with low SRP/TP ratios and a second grouping of clay basins with moderate ratios.

The illusion of a slight negative trend with area for total nitrogen (Fig. 7A) may arise from a clustering of the smallest basins representing subthreshold areas as noted above. The marginally higher loadings associated with clay basins may indicate relationships amongst crop type, farming practices, overland flow and particulate-dependency. These annual values may camouflage pronounced seasonal trends associated with flushing of nitrogen during the spring. Organic nitrogen data mirror those for total nitrogen. Nitrate nitrogen, the most common of the nitrogen species, is neither related to area nor consistently to overburden texture.

Chloride

The mean annual loading of chloride is approximately 80,000 metric tons (9.4 tons per square kilometer or 54 lb per acre). Chloride is found in lower concentration in various rock types than any other major constituent of natural water (Hem 1970). In the Lake Erie watershed, bedrock is generally covered to considerable depth with late-glacial freshwater sands and clays. Limited outcrops of Devonian and Silurian dolomite in the lower and middle reaches of the Grand River watershed may provide a limited source of chloride. The significance of chloride in natural waters has been summarized in Hem (1970).

> The chemical behaviour of chloride in natural water is very tame and subdued compared to the other major ions. Chloride ions do not significantly enter into oxidation or reduction reactions, form no important solute complexes with other ions, do not form salts of low solubility, are not significantly adsorbed on mineral surfaces, and play few vital biochemical roles. The circulation of chloride ions in the hydrologic cycle is largely through physical processes.

Although chloride may pulse through watersheds at different rates depending upon terrain and hydrologic variables, representative annual loadings might be expected to be unrelated to overburden texture and should exhibit relatively uniform unit yields between basins. The present data (Figure 7B) exhibit both a slight differentiation on the basis of texture and a negative relationship between unit yield and basin area (excepting basins #65, 67, 68). The latter observation is intriguing in view of the conservative nature of the chloride ion and suggests chloride storage proportional to basin area. Hem (1970) concluded that a few tenths of a milligram of cyclic chloride per litre of rainfall are insufficient to account for observed concentration in runoff. In Ontario, chloride compounds widely used for road de-icing may account for the observed loadings. The observation that the three lowest values per square kilometer (#65, 67, 68) have significantly fewer linear kilometers of improved

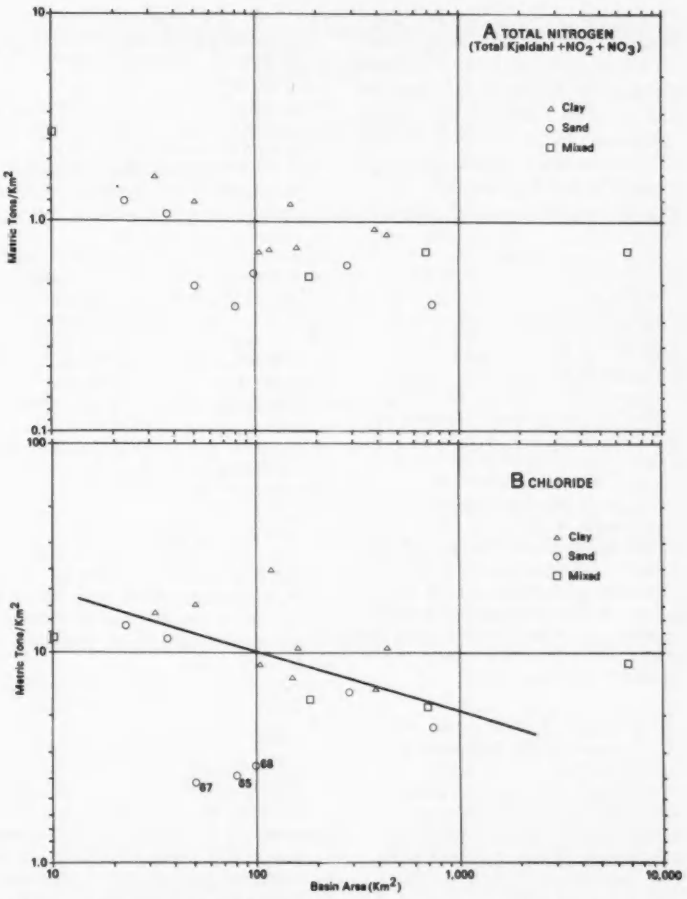

FIG. 7. Mean annual yield of A) total nitrogen and B) chloride against basin area.

road than any other tributary basin suggests that a correlation between chloride loadings and length of improved road per basin might be instructive.

Summary and Conclusions

Federal discharge records and concentration data for solids, nutrients and chloride, collected on a routine basis over the period 1964 through 1972 by the Ontario Ministry of the Environment have been utilized to calculate loadings for nineteen Canadian river mouth locations on Lake Erie. In view of the stage-dependencies known to exist for certain parameters, the bias of concentration data towards low flow conditions causes serious undercalculation of loadings. Nevertheless, the loadings represent the best available information to date. Admitting data inadequacies, the unit yields are informative for heuristic purposes. A storage effect produced by increasingly longer and more complex flow paths is apparent for some suspended solids, particulate-bound nutrient and chloride. Not surprisingly, yield values for these data do not readily cluster by basin substrate texture (sand/clay); an exception is the plot of SRP as a function of TP. Basin yields of SRP are shown to be closely associated to TP yields, at least under the low flow conditions represented herein. The data for SRP versus TP and for total nitrogen suggest a minimum threshold basin size below which nutrient runoff is relatively unaffected by basin storage and/or uptake associated with longer temporal and spatial flow paths of larger basins. The threshold area, in the

order of 40 km², is well below the threshold value of 250 km², beyond which Slaymaker and McPherson (1973) claim that the land use variable is of minor importance. Although the positive relationship between chloride loadings and basin area suggests that chloride can be related to length of improved road, the negative trend between chloride yield per square kilometer and basin area indicates a storage function proportional to basin size. This latter observation is inconsistent with the conservative nature of the chloride ion.

Acknowledgments

The study was carried out as part of the Task D activities of the Pollution from Land Use Activities Reference Group, International Joint Commission, and was funded by Environment Canada through the Canada Centre for Inland Waters (C.C.I.W.). Publication of this report is with permission of the Reference Group but does not necessarily reflect the Group's views. Certain data files employed in this report were generously made available by the Water Survey of Canada and the Ontario Ministry of the Environment. I gratefully acknowledge the collaboration of Dr R. L. Thomas, Mr J. Ralston and the useful comments of colleagues at C.C.I.W. Several referees made valuable suggestions. The Department of Geography at Queen's University made space available to house the project.

ARMSTRONG, D. E., K. W. LEE, P. D. UTTORMARK, D. R. KEENEY, AND R. F. HARRIS. 1974. Pollution of the Great Lakes by nutrients from agricultural land, *In* PLUARG-Task A, Management programs, effects of research and present land use activities on water quality of the Great Lakes. U.S. Task A report to Pollution from Land Use Activities Reference Group of the International Joint Commission, Section 6, Vol. 1, 108 p.

BRUNE as reported in LANGBEIN, W. B., AND S. A. SCHUMM. 1968. Yield of sediment in relation to mean annual precipitation. Trans. Am. Geophys. Union 39: 1076–1084.

BURNS, N. M. 1976. Nutrient budgets for Lake Erie 1970. J. Fish. Res. Board Can. 33: 520–536.

CAHILL, T. H., P. IMPERATO, P. K. NEBEL, AND F. H. VERHOFF. 1974. Phosphorus dynamics in a natural river system. American Institute of Chemical Engineers Symposium Series No. 145, Vol. 1, New York, N.Y.

CARTER, D. L., J. A. BONDURANT, AND C. W. ROBBINS. 1971. Water-soluble NO_3—N, PO_4—P, and total salt balances on a large irrigation tract. Soil Sci. Soc. Am. Proc. 35: 331–335.

ENVIRONMENTAL PROTECTION AGENCY. 1971. Agricultural pollution of the Great Lakes basins. U.S. Government Printing Office, Washington, D.C.

FOURNIER, F. 1960. Climat et erosion. Presses Universitaires de France, Paris. 201 p.

GREAT LAKES WATER QUALITY BOARD. 1973. Annual report to the International Joint Commission.

HEM, J. D. 1970. Study and interpretation of the chemical characteristics of natural water (second edition). U.S. Geological Survey Water-Supply Paper #1473, Washington, D.C.

INTERNATIONAL JOINT COMMISSION. 1969. Pollution of Lake Erie, Lake Ontario and the international section of the St. Lawrence River, Vol. 2 — Lake Erie. Report to the International Joint Commission by the International Lake Erie Water Pollution Board, and the International Lake Ontario–St. Lawrence Water Pollution Board.

JOHNSON, W. R., F. ITTIHADICH, R. M. DAUM, AND A. F. PILLSBURY. 1965. Nitrogen and phosphorus in tile drain effluent. Soil Sci. Soc. Am. Proc. 29: 287–289.

KEMP, A. L., ET AL. 1974. Sedimentation rates and recent sediment history of Lakes Ontario, Erie and Huron. J. Sediment. Petrol. 44: 207–218.

MACCRIMMON, H. R., AND M. G. JOHNSON. 1971. Limnology of the lower Grand River. For C.C.I.W., Burlington, *Cited in* MacLean, A. J., and E. M. MacDonald. 1973. Agricultural contributions to nutrient enrichment of waters in watersheds of Lake Erie, Lake Ontario and the international section of the St. Lawrence River. Part of a report of the Canada Department of Agriculture Task Force for Implementation of Great Lakes Water Quality Program.

NEUHOLD, B. N. 1975. The impact of urban development on the hydrogeomorphology of Highland Creek drainage basin, Ontario. Unpublished M.Sc. Thesis. Department of Geography, Queen's University, Kingston, Ont. 148 p.

ONGLEY, E. D. 1973. Sediment discharge from Canadian basins into Lake Ontario. Can. J. Earth Sci. 10: 146–156.

1974. Hydrophysical characteristics of Great Lakes tributary drainage, Canada. Report for Task D, Pollution from Land Use Activities Reference Group, International Joint Commission, 5 vol.

PLUARG-TASK A. 1974. Management programs, effects of research and present land use activities on water quality of the Great Lakes. U.S. Task A report to Pollution from Land Use Activities Reference Group of the International Joint Commission.

SLAYMAKER, O. H., AND H. J. MCPHERSON. 1973. Effects of land use on sediment production, p. 158–183. *In* Fluvial processes and sedimentation, Proceedings of Hydrology Symposium, University of Alberta, National Research Council of Canada.

SLY, P. G. 1976. Lake Erie and its basin. J. Fish. Res. Board Can. 33: 355–370.

STICHLING, W. 1973. Sediment loads in Canadian rivers, p. 38–72. *In* Fluvial processes and sedimentation, Proceedings of Hydrology Symposium, University of Alberta, National Research Council of Canada.

1974. Sediment loads in Canadian rivers. Inland Waters Dir. Tech. Bull. 74: 27 p.

TERRY, R. D., AND S. E. SALBACH. 1969. Land use nutrient level relationships on two Southern Ontario watersheds. Can. Sym. Water Pollut. Res. 5: 171–194.

WEIBEL, S. R. 1969. Urban drainage as a factor in eutrophication, p. 383–403. *In* Eutrophication: causes, consequences, correctives. National Academy of Sciences, Washington, D.C.

Temperature, Oxygen, and Nutrient Distribution Patterns in Lake Erie, 1970

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BURNS, N. M. 1976. Temperature, oxygen, and nutrient distribution patterns in Lake Erie, 1970. J. Fish. Res. Board Can. 33: 485–511.

Ten major surveys of biological and chemical variables in Lake Erie were carried out during the 1970 shipping season. The observed distributions of temperature, turbidity, oxygen, soluble and particulate phosphorus, nitrate plus nitrite, ammonia, and soluble reactive silica are presented. Comment on the probable causes of the observed distribution patterns has been included where appropriate. Ammonia quantities in the lake remained almost unchanged throughout the year while nitrate plus nitrite quantities varied considerably. The ratio of particulate to total phosphorus showed very little seasonal change.

BURNS, N. M. 1976. Temperature, oxygen, and nutrient distribution patterns in Lake Erie, 1970. J. Fish. Res. Board Can. 33: 485–511.

On a fait 10 relevés majeurs des variables biologiques et chimiques dans le lac Erié durant la saison de navigation de 1970. Nous présentons ici les observations sur la répartition de la température, turbidité, oxygène, phosphore soluble et particulaire, nitrate plus nitrite, ammoniac et silice réactive soluble. Nous faisons, s'il y a lieu, des commentaires sur les causes probables des répartitions observées. Les quantités d'ammoniac dans le lac demeurent presque constantes durant toute l'année, alors que celles de nitrate plus nitrite varient considérablement. Le rapport phosphore particulaire: phosphore total montre très peu de changement saisonnier.

Accepted November 18, 1975 Accepté le 18 novembre 1975

THE more important lakewide studies to the present, which give a general outline of the chemistry of Lake Erie, are by Kramer (1961), Weiler and Chawla (1968), and Gächter et al. (1974). A more recent large-scale investigation consisting of 10 major surveys of the lake during 1970 for the measurement of basic chemical and biological parameters is reported here. The data from the surveys on the chemical variables are extensive and, after processing, have resulted in approximately 80 distribution maps and tables, some of which are published here. The full data set is published elsewhere (Burns et al. 1976a). This report describes the variations observed in the more important parameters in 1970, together with possible explanations for the observed variations.

Sampling and Data Processing Methods

The sampling network pattern of 56 station locations (Fig. 1) was based on subjective judgements (from the results of previous surveys) as to what would constitute adequate coverage of the lake.

At all stations, a bathythermograph trace was obtained and water was sampled using Van Dorn bottles at the following depths: 1 and 5 m below surface, 2 m above the bottom, plus two samples spaced equally (if possible) between the 5-m below surface and 2-m above bottom sampling depths. Single determinations were made on each sample for the following parameters: oxygen concentration, conductivity, turbidity, concentration of soluble reactive phosphorus, NO_3^- plus NO_2^-, NH_3, soluble reactive silica, pH, alkalinity (filtered), and total phosphorus (filtered and unfiltered). The samples taken at 1.0 m from the surface and 2.0 m from the bottom were filtered and analyzed for Cl^-, SO_4^{2-}, K^+, Na^+, Ca^{2+}, and on three selected cruises the samples were further analyzed for Li^+, filtered iron, and filtered manganese. The analytical methods used in the various

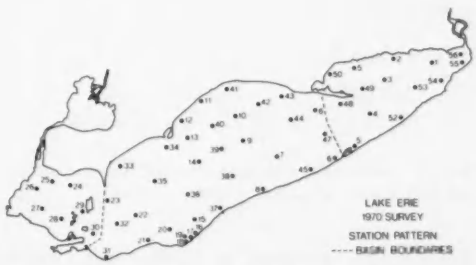

FIG. 1. Lake Erie 1970 Survey Station Pattern showing basin boundaries.

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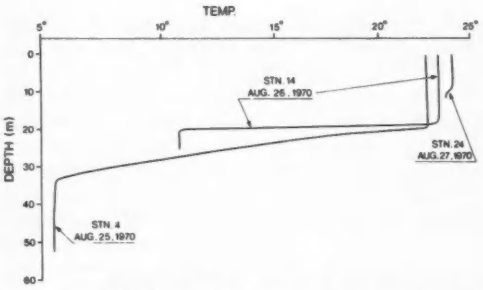
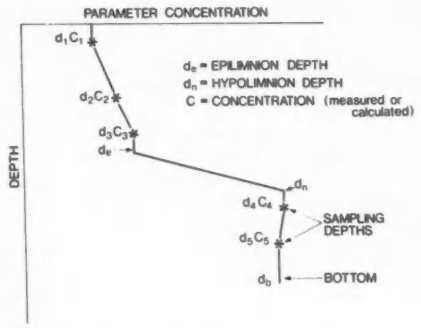


FIG. 2. Bathythermograph traces typical of those observed in Lake Erie during the stratified period.



$$C_{EPILIMNION} = \frac{1}{d_e}\left(d_1C_1 + (d_2-d_1)\times\frac{1}{2}(C_1+C_2) + (d_3-d_2)\times\frac{1}{2}(C_2+C_3) + (d_e-d_3)C_3\right)$$

$$C_{MESOLIMNION} = \frac{1}{2}(C_3+C_4)$$

$$C_{HYPOLIMNION} = \frac{1}{(d_b-d_n)}\left((d_4-d_n)C_4 + (d_5-d_4)\times\frac{1}{2}(C_4+C_5) + (d_b-d_5)C_5\right)$$

FIG. 3. Schematic representation of the method for the calculation of depth-weighted averages.

determinations are given by Philbert and Traversy (1973). Estimates of the precision for each parameter, with the errors of both the analytical method and sampling from the ship taken into account, are given by Strachan (1973).

Typical bathythermograph traces for the three basins are shown in Fig. 2. It can easily be seen that the traces for the Central and Eastern basins are close to being ideal, with both the epilimnion and hypolimnion being isothermal and the thermocline temperature change being linear with depth. Further, the bathythermograph traces showed that the water temperature at a station was usually constant from the surface to the bottom of the lake in the spring and fall. During Project Hypo (Burns and Ross 1972), when stations were occupied and reoccupied on a 4-day basis, it was observed that any secondary thermal layering in the epilimnion or hypolimnion had an existence of less than 4 days.

It was decided to examine the water masses in accordance with their thermal characteristics, that is, as individual, well-mixed isothermal layers. If the lake was found to be isothermal with depth it was considered to consist of only one layer, which for the purpose of the present discussion is called the epilimnion. Under stratified conditions, the lake was considered to consist of three layers, with the epilimnion and hypolimnion being well-mixed layers and the mesolimnion being a pycnocline preventing direct mixing between the upper and lower water masses. It has been shown that a near-linear chemocline exists in Lake Erie coinciding with the pycnocline (Burns and Ross 1972). The vertical position of the chemocline was determined for each station by examining the bathythermograph trace and determining the depths of the temperature discontinuities for that station. The depth-weighted mean concentration for each layer at each station was determined using the calculation scheme shown in Fig. 3.

Epilimnion and hypolimnion distribution maps were drawn using the calculated depth-weighted concentration values for the 56 observation stations and are shown below.

Results and Discussion

TEMPERATURE

Two features are noticeable in the hypolimnion temperature maps, Fig. 4; the first being the apparent leakage of Eastern basin hypolimnion water into the Central basin prior to Survey 3 as shown by the 6 C isotherm and survey 4 as shown by the 7 C isotherm. The second feature is the gradual crowding of the coldest hypolimnion water into the northwest corner of the Central basin. The reason for this is that the hypolimnion currents flow predominantly to this quarter (Simons 1976; Hartley 1968) and probably cause hypolimnion upwelling along the northwest shore of the Central basin.

The surface temperature distributions are shown in Fig. 5. The first major surface temperature phenomenon was a warming trend which began in April in the west and shallow shoreline areas, with maximum overall temperatures being reached at the end of August. A cooling trend then started with the remaining highest December temperatures being observed in the center of the Eastern basin.

TURBIDITY

The turbidity of lake water is caused primarily by phytoplankton, organic detritus, and suspended inorganic material (SIM). The inorganic material usually consists of clays, either resuspended from the bottom or brought in by rivers

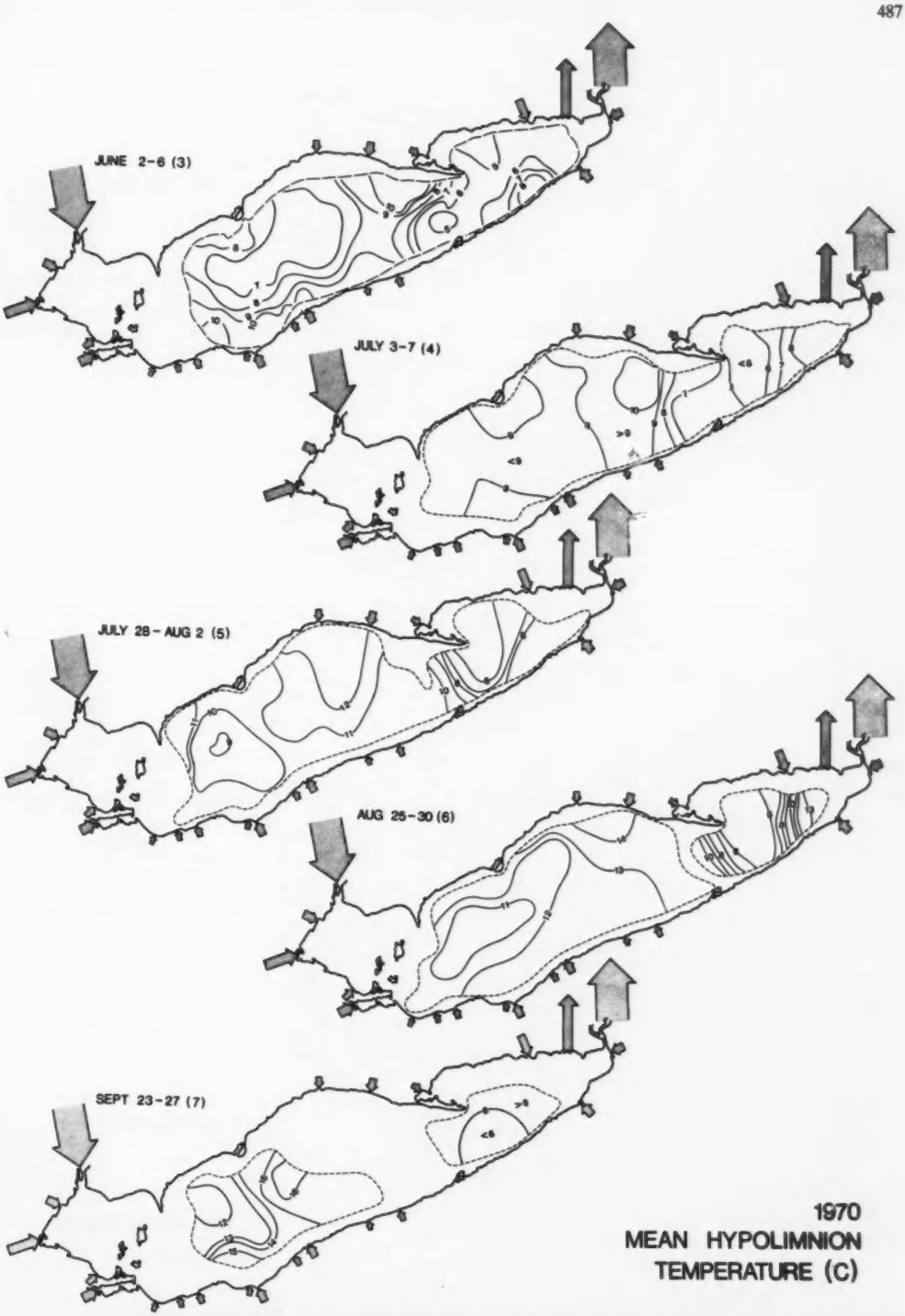


FIG. 4. Lake Erie mean hypolimnion temperatures (June–September 1970).

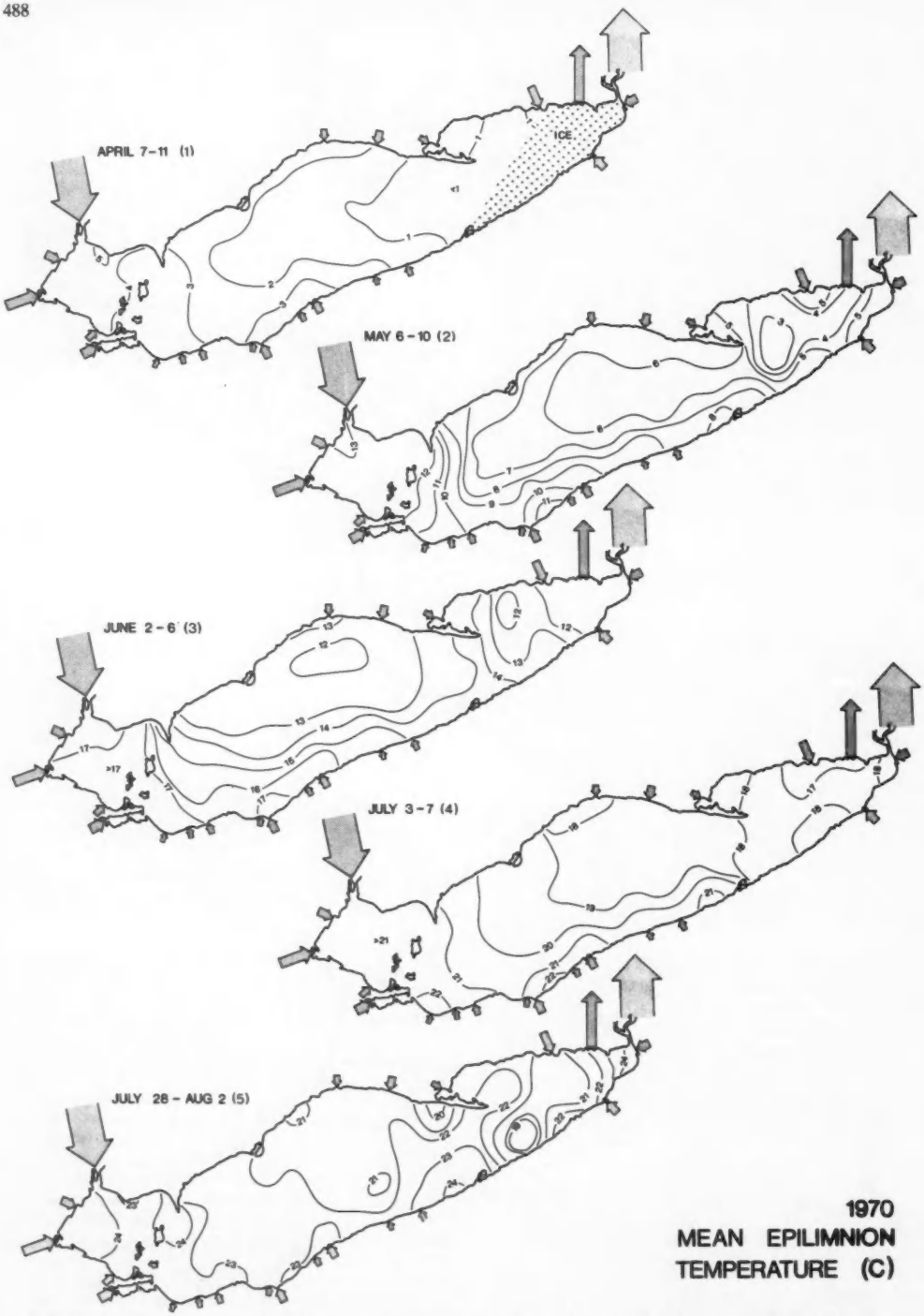


FIG. 5. Lake Erie mean epilimnion temperatures (April–December 1970). The arrows depict the positions of water flow in and out of the lake with the width of the arrows being proportional to the average water flows. Survey numbers are noted in brackets.

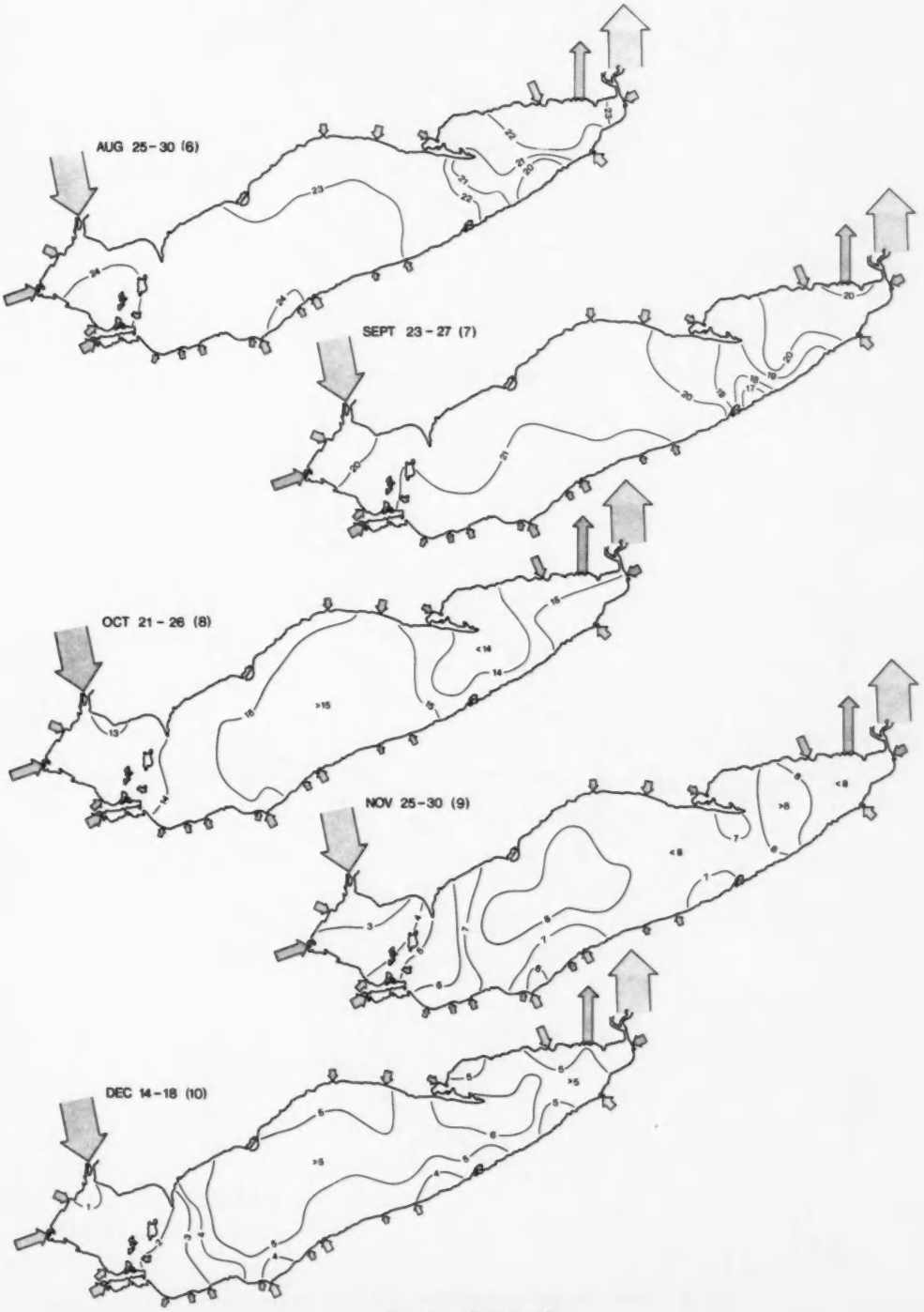


FIG. 5. *(Continued).*

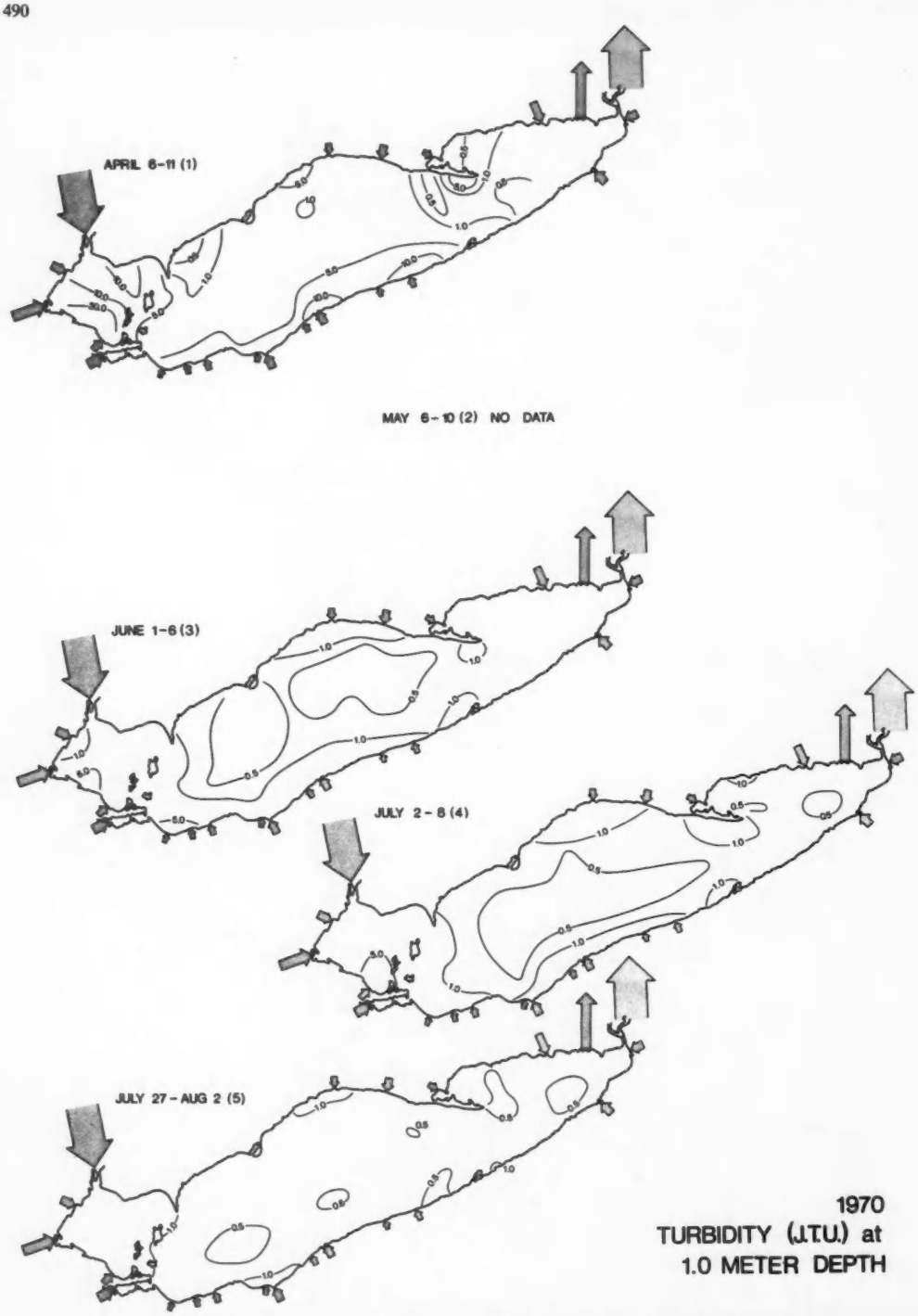


FIG. 6. Lake Erie surface turbidities (April–December 1970).

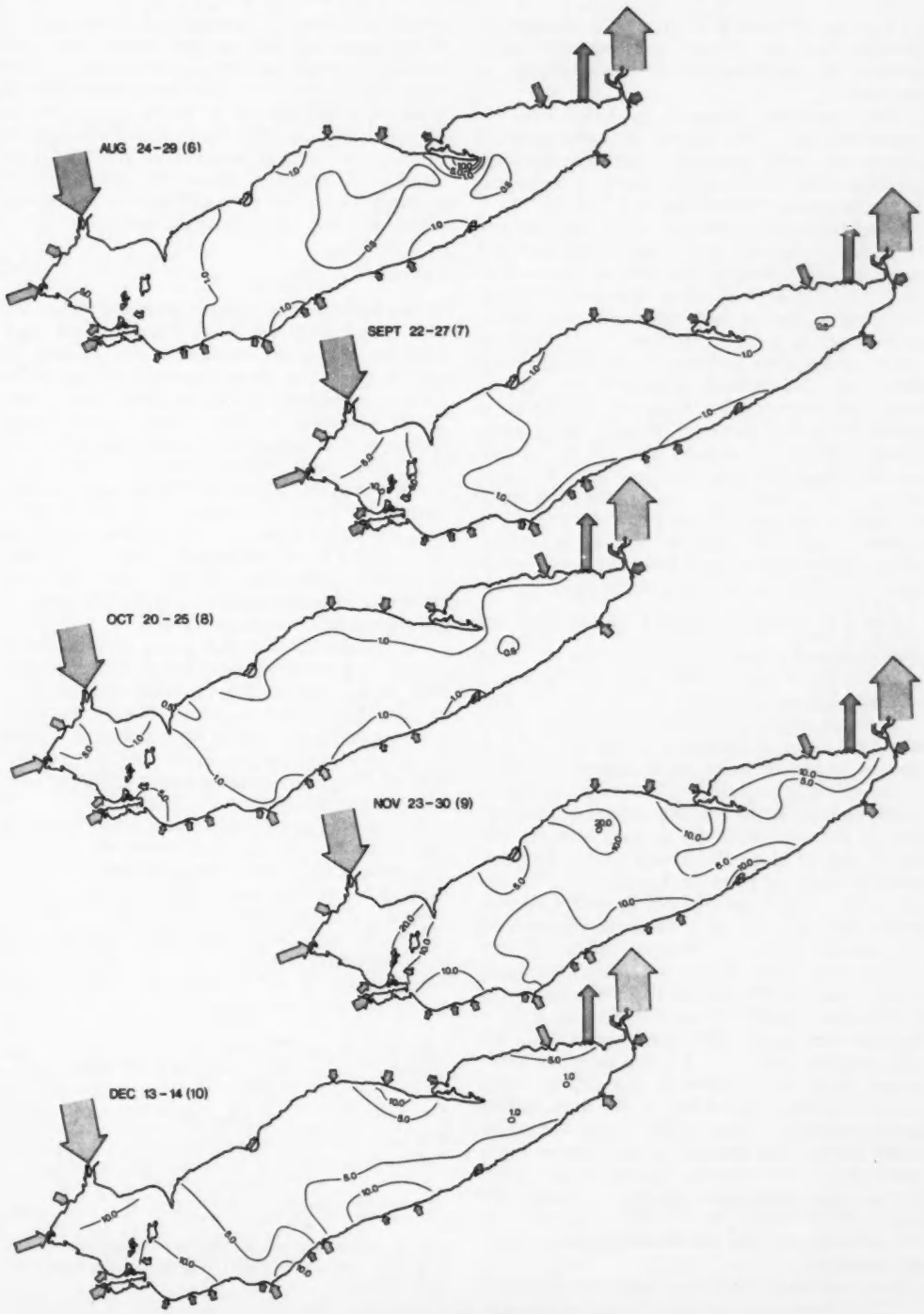


FIG. 6. *(Continued).*

or by shore erosion. It is of interest to know, if possible, how the organic and inorganic components can each separately affect the turbidity of the water.

The turbidities observed in Lake Erie are shown in Fig. 6. The interval between survey 4 and survey 5 was generally calm with the only prolonged periods of winds above 15 knots experienced being between July 14 and 16 (T. J. Simons personal communication). Thus, it can be assumed that very little resuspended sediment material was present in the water at the time of survey 5. The phytoplankton biomass density in the Western basin water was close to the summer average of 4 g m^{-3} (Munawar and Munawar 1976) yet the mean turbidity in the Western basin was only 1.3 Jackson Turbidity Units (J.T.U.) with a maximum observed value of 2.4 J.T.U. at station 25 during survey 5. Thus, it seems probable that turbidities in Lake Erie above 3 J.T.U. are not caused by high phytoplankton densities unless circumstances are exceptional. The cause of high turbidities (values above 5 J.T.U.) is almost certainly the presence of SIM in the lake water. Data collected on a survey of Lake Erie in April 1973 gave the following relationship:

$$\text{SIM (g m}^{-3}) = 0.92 \times \text{Turbidity (J.T.U.)} + .02 \quad (1)$$

This regression relationship is based on 239 pairs of data from 1 to 45 J.T.U. covering the whole lake and having a correlation coefficient of 0.92.

It would seem reasonable to use the SIM–turbidity relation to obtain some idea of the suspended inorganic material in the water under conditions of high turbidity and even to make a rough estimate of the resulting sediment transport. During the stormy 19-day interval between surveys 9 and 10 (Lam and Jaquet 1976), the mean turbidity south of Pelee Pt. (stations 23, 32) was 8.4 J.T.U., which corresponds to a SIM content in the water of 7.7 g m^{-3}. From flows estimated by Simons (1976), the transport of fine-grained sediment by water during this period from the Western basin to the Central basin was calculated to be approximately 73×10^3 m tons. If it is assumed that similar turbulent periods exist for one-fourth of each year, then the sediment transported from the Western to the Central basin would be in the order of 0.34×10^6 m tons. This output quantity is small (8%) when compared to the annual sedimentation in the Western basin (about 4.0×10^6 m tons), (Kemp et al. 1976). Further, this observation tends to confirm the conclusion of Kemp et al. (1976) that little of the sediment entering the Western basin is actually leaving it.

It is considered likely that turbulent conditions would cause vertical concentration gradients of resuspended materials in the water with high turbidities occurring close to the bottom of the lake. Figure 7 shows a vertical cross-section of observed turbidities. It is worth noting that at distances of more than 2.0 m above the bottom, there is no vertical gradient of turbidity even when a horizontal gradient exists. If a vertical gradient does exist, it would have to be in the region less than 2.0 m above the bottom.

Oxygen

The hypolimnion oxygen saturation values are shown in Fig. 8. The main feature of the hypolimnion oxygen distributions is the increase in oxygen saturation going from west to east in the Central basin and the comparatively low oxygen values observed in the Eastern basin during survey 6. The oxygen saturation values of less than 30% in the Eastern basin are largely a result of the thinness of the hypolimnion in that part of the basin. By survey 7 the mesolimnion had descended such that the shallow peripheral part of the hypolimnion had been incorporated into the mesolimnion and the water containing the lower concentrations of oxygen had been incorporated into the mesolimnion.

A detailed study of the causes and results of anoxic conditions in the Central basin hypolimnion during August 1970 has been reported on in Project Hypo (Burns and Ross 1972). The accurate calculation of oxygen depletion rates is complex and is discussed by Burns (1976a) together with an estimation of the monthly depletion rates for the Central and Eastern basins.

The epilimnion oxygen conditions are shown in Fig. 9. These maps give indications of the times and areas of the phytoplankton growth periods in the lake by showing the areas of high percentage oxygen saturation. The major growth periods appear to have been observed in the western basin during surveys 2 and 6, in the Central basin during surveys 2 and 3, and in the Eastern basin during survey 3. An examination of the particu-

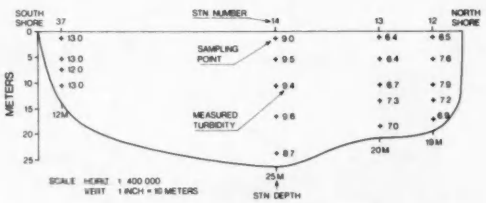


FIG. 7. Cross-section of observed turbidities in Lake Erie (November 27, 1970). (Units are Jackson Turbidity Units.)

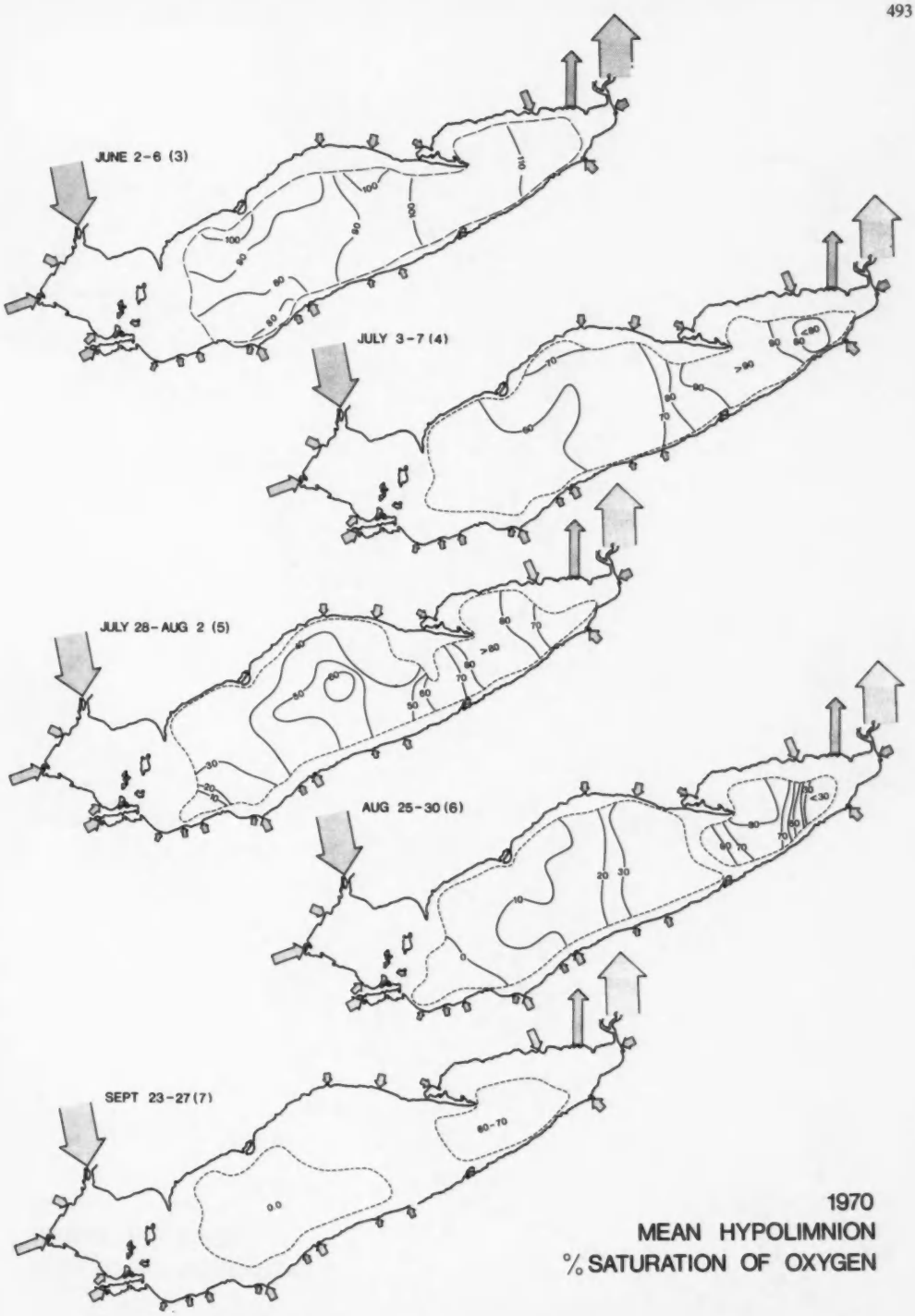


FIG. 8. Lake Erie mean hypolimnion oxygen percentage saturation (June–September 1970).

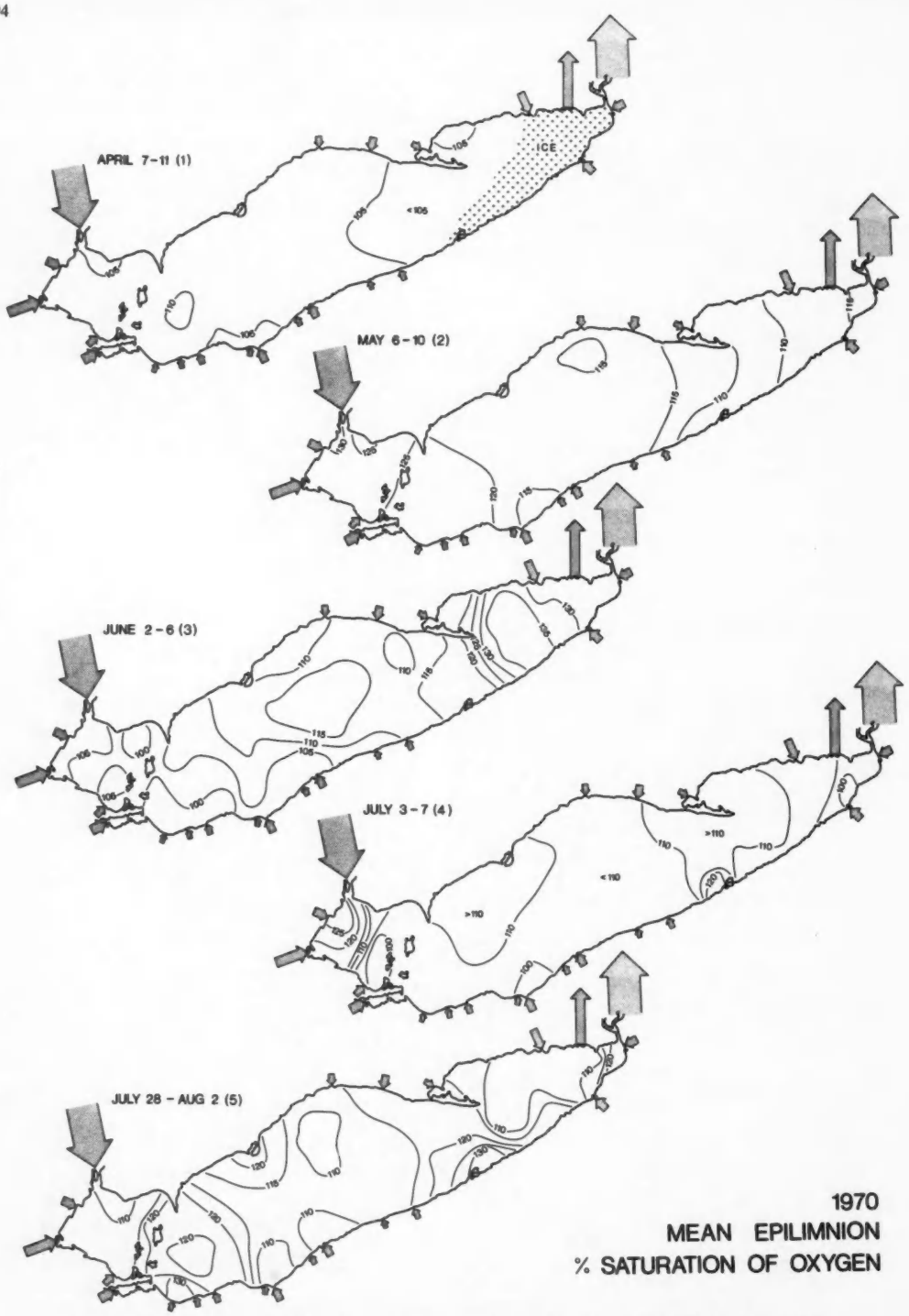


FIG. 9. Lake Erie mean epilimnion oxygen percentage saturation (April–December 1970).

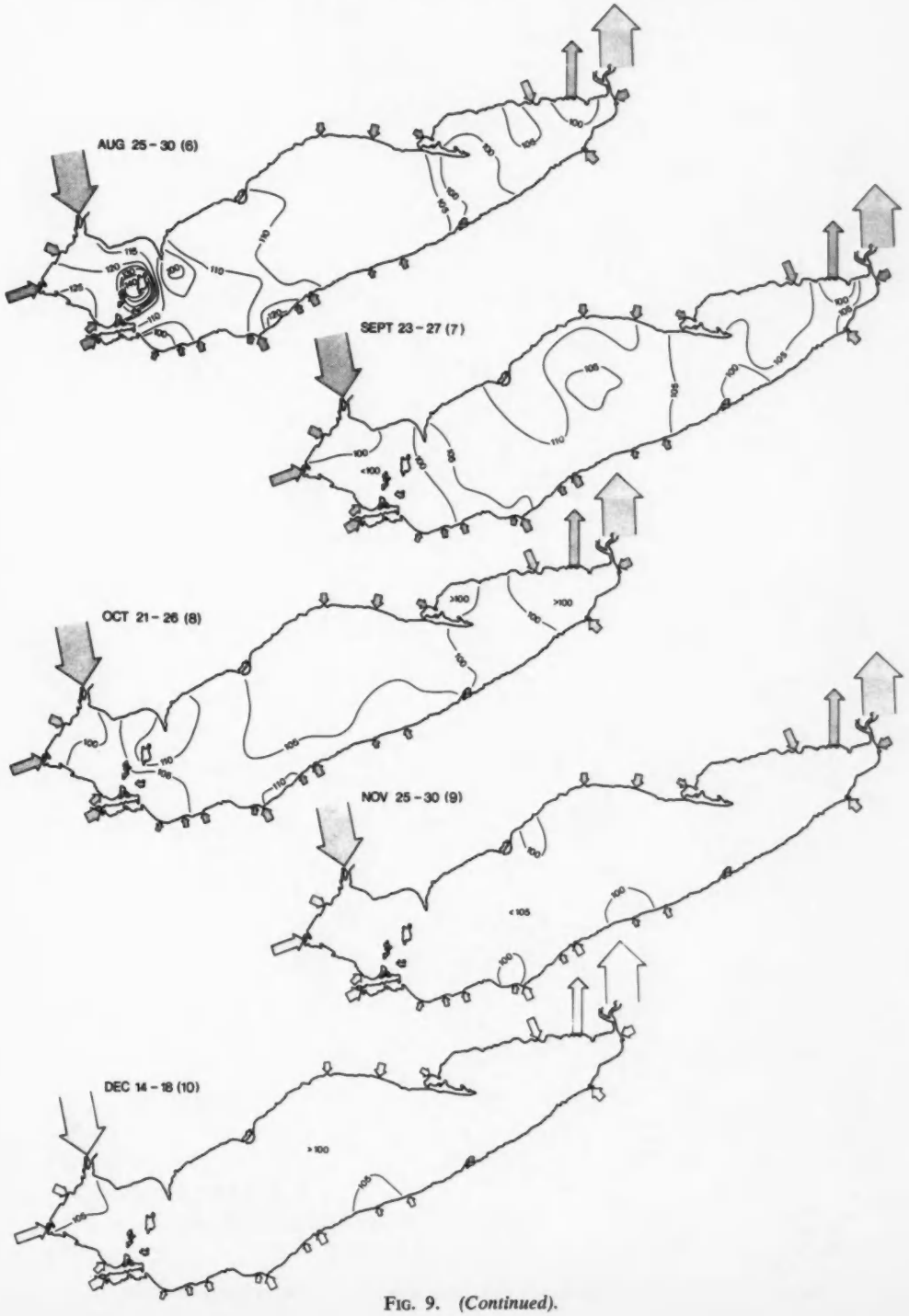


FIG. 9. *(Continued).*

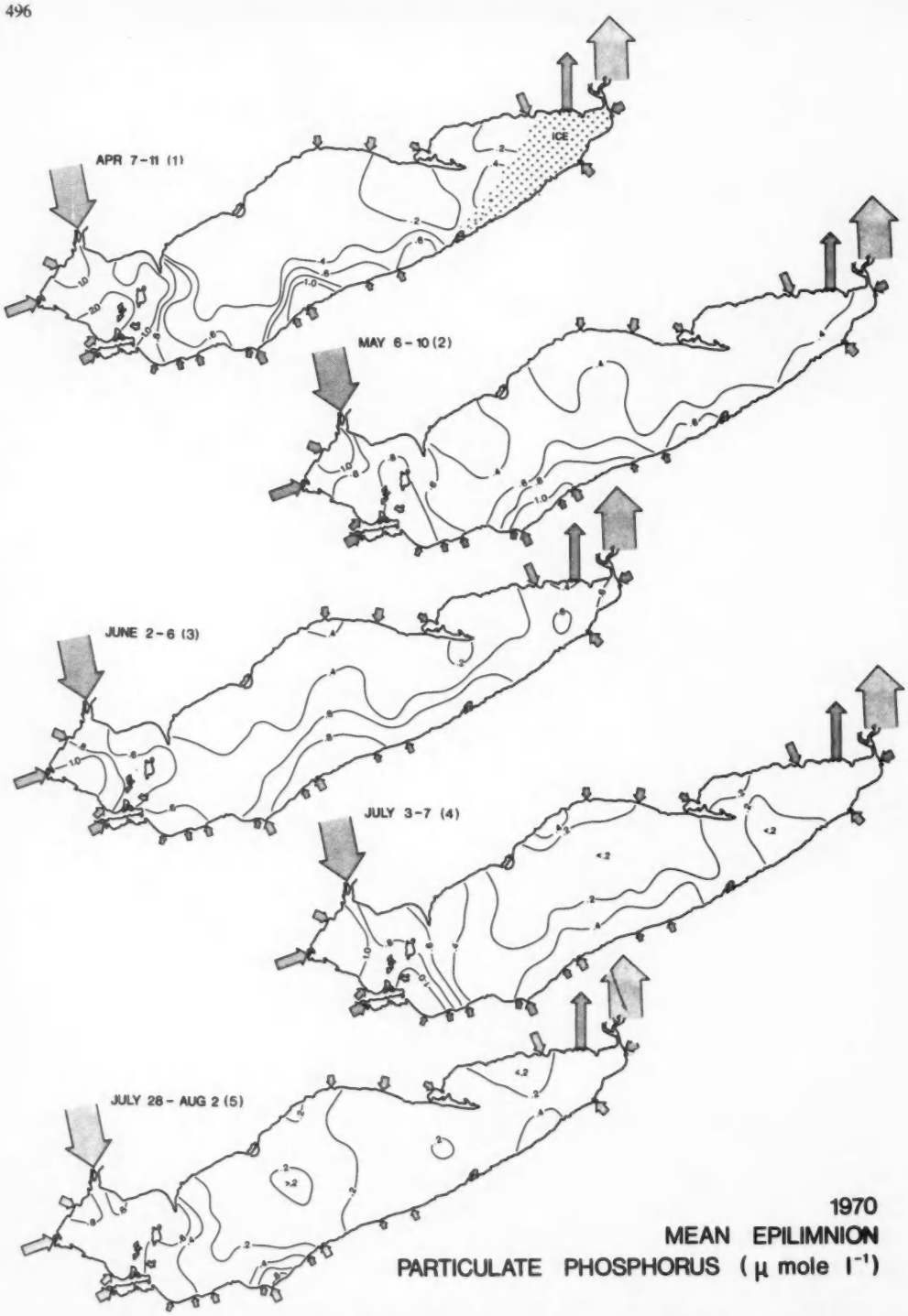


FIG. 10. Lake Erie mean epilimnion particulate phosphorus (April–December 1970).

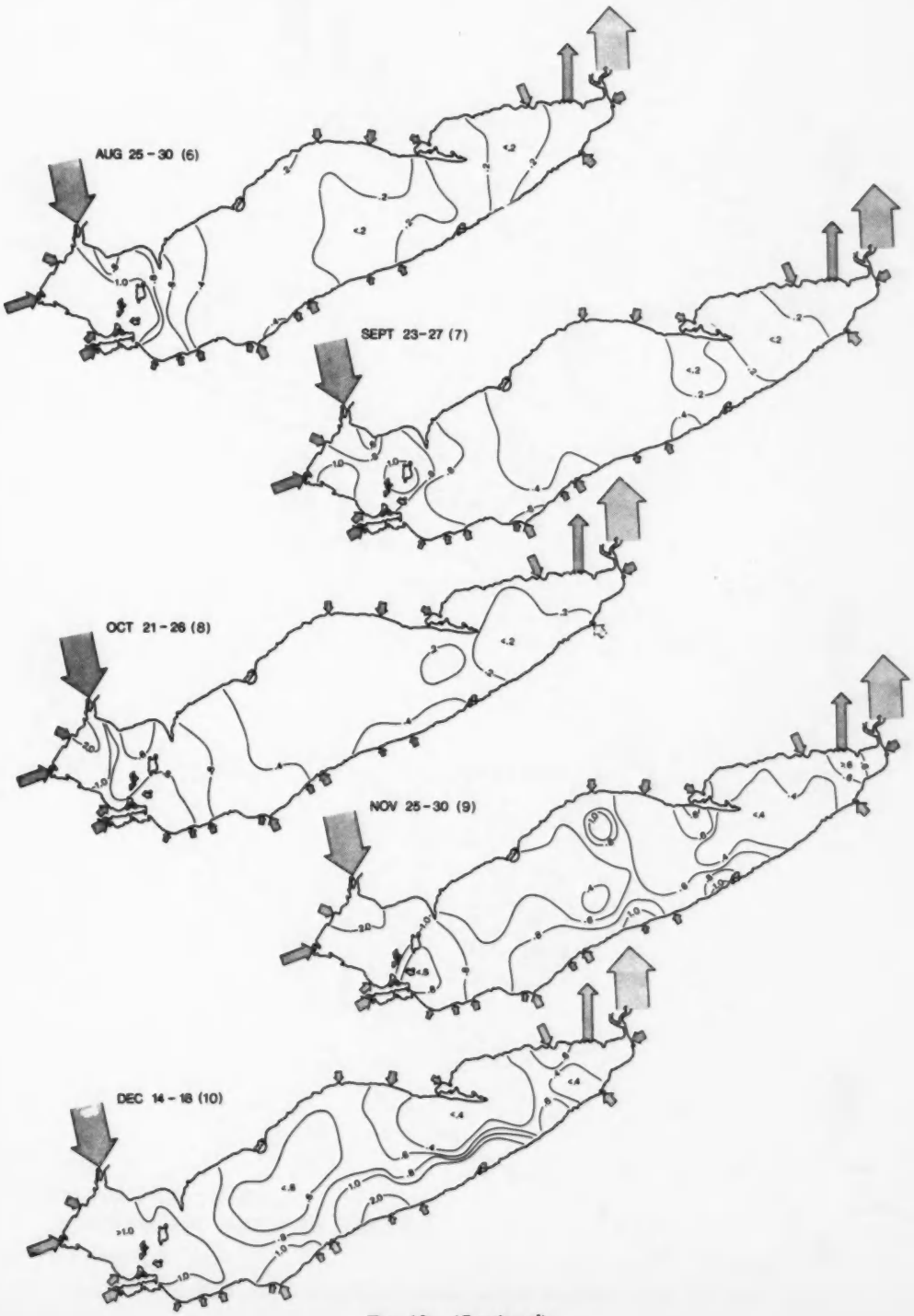


FIG. 10. *(Continued).*

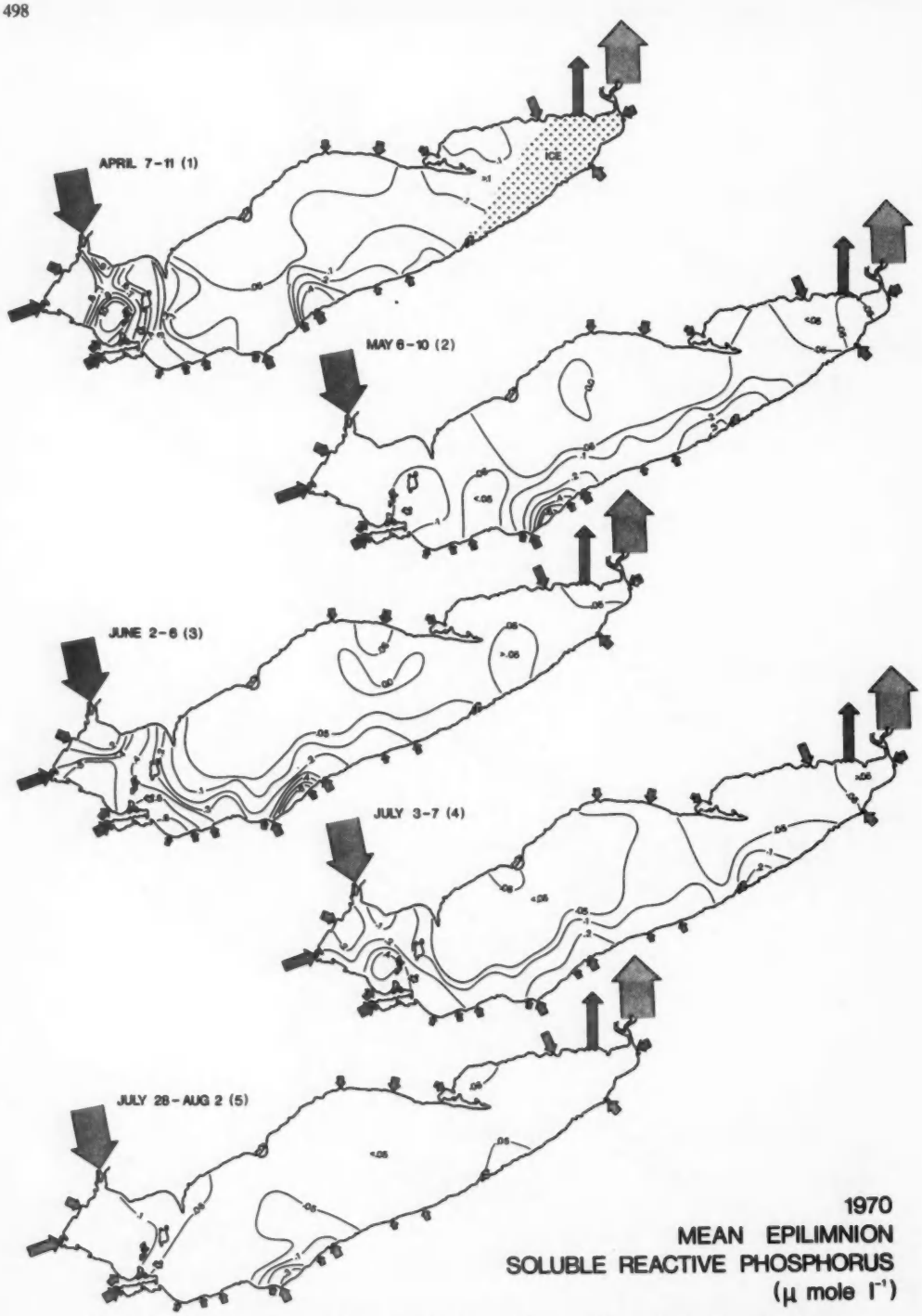


FIG. 11. Lake Erie mean epilimnion soluble reactive phosphorus (April–December 1970).

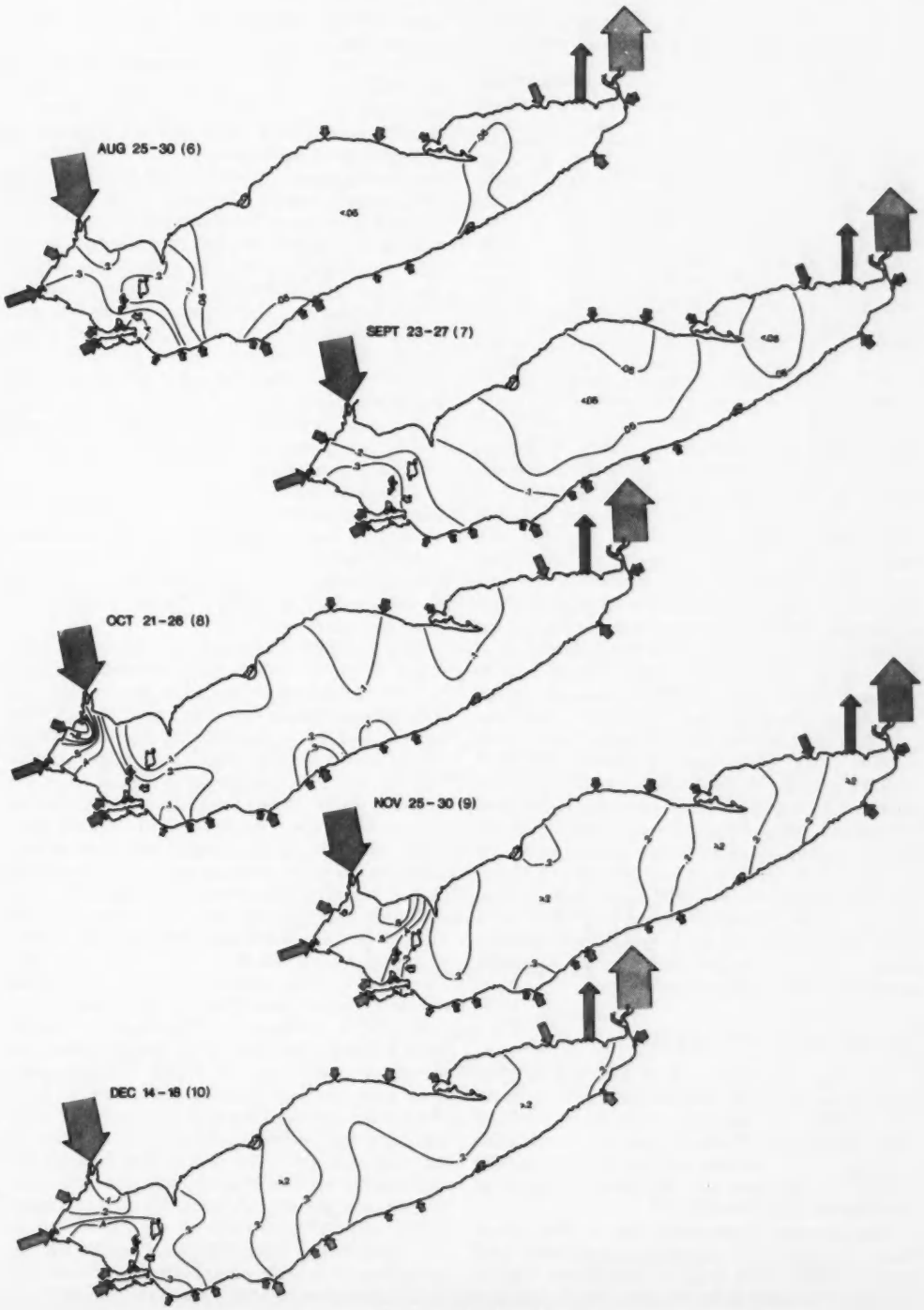


FIG. 11. *(Continued).*

TABLE 1. Ratios of particulate phosphorus to total phosphorus as observed in Lake Erie epilimnia, 1970.

Date	Survey No.	Western basin	Central basin	Eastern basin
Apr. 7–11	1	0.63	0.59	0.44
May 6–10	2	0.75	0.67	0.63
June 2–6	3	0.58	0.69	0.72
July 3–7	4	0.69	0.54	0.44
July 28–Aug. 2	5	0.68	0.54	0.64
Aug. 24–30	6	0.65	0.67	0.68
Sept. 23–27	7	0.62	0.64	0.56
Oct. 21–26	8	0.66	0.58	0.44
Nov. 25–30	9	0.65	0.59	0.51
Dec. 14–18	10	0.67	0.63	0.58
	Avg	0.66	0.61	0.56
	SD	0.05	0.05	0.10

late to total phosphorus ratios (Table 1) reveals that the highest values for the ratio were observed in the Western basin during survey 2, in the Central basin during surveys 2 and 3, and in the Eastern basin during survey 3. Also, in each case, the high particulate to total phosphorus ratio was followed in the subsequent survey by the lowest value of the ratio for the basin. Thus, it appears that the major growth period in the Western basin was in early May followed by a period when much of the newly formed organic material settled out of the water. A similar growth period occurred during May in the Central basin and in late May and early June in the Eastern basin. These growth periods coincided with the onset of thermal stratification in the two basins. The subsequent low particulate to total phosphorus ratios were probably a result of the loss of organic material which settled out of the comparatively shallow, newly formed epilimnia into the hypolimnia. It would seem that although the particulate to total phosphorus ratio is a fairly stable value for each basin, small changes in the value can give an indication of the major occurrences of phytoplankton growth.

VARIOUS FORMS OF PHOSPHORUS

The diagrams in Fig. 10–12 give the distributions of some of the phosphorus forms. Particulate phosphorus constitutes from 50 to 70% of total phosphorus (Table 1) and the distribution patterns of both are very similar. Soluble reactive phosphorus demonstrates significant variation in the lakewide distributions.

The inputs of rivers and cities to the Central basin are almost all southshore inputs and total approximately 5900 tons of phosphorus (Burns 1976b). The northshore inputs from the eroded cliffs deposit approximately 14,000 m tons of phosphorus annually into the lake (Burns et al. 1976b). The phosphorus distribution maps demonstrate quite clearly that the smaller amounts of anthropogenic phosphorus originating from the south shore have a much greater effect on the phosphorus distributions in the lake than the larger deposition of phosphorus from the cliffs. This pattern can be seen particularly in the case of soluble reactive phosphorus which is the most biologically active form and thus has considerable effect on water quality. The phosphorus entering the lake from the north shore is composed of 95% apatite and the evidence from the phosphorus distribution maps is that these northshore apatite inputs settle out quickly and play almost no part in the phosphorus cycle of the lake. This is understandable since the apatite is dense (sp gr = 3.1–3.2), relatively coarse grained (size range 5–100 μm) and very insoluble (J. D. Williams personal communication).

The hypolimnion waters of the Central basin showed a dramatic increase in the concentration of soluble reactive phosphorus when the waters became anoxic (Fig. 8 and 12). The concentrations increased with time because of the continued loading of phosphorus to anoxic water from the sediments.

The standard deviations of the mean basin ratios of particulate to total phosphorus (Table 1) are smaller than expected, considering that the summer biomass levels can be at least 3 times greater than the spring and fall levels (Munawar and Munawar 1976). The near constant particulate to total phosphorus ratio could be maintained if the biomass increase is accompanied by a decrease in the concentration of organic detritus in the water during the summer. This replacement of organic detrital carbon by biomass carbon has been observed by Leach (1975) who estimated that during the years 1971–73, the ratio of detritus to phytoplankton organic carbon decreased from a value of 3:1 in the spring to a value of 1:1 in the summer in the Western basin; the ratio returned to a value of 3:1 in the fall.

As might be expected, the shallow Western basin has a greater fraction of the phosphorus in particulate form than the deeper Eastern basin. It is worth noting that the particulate to total phosphorus ratios as calculated for survey 9 data, just after the November storm, are all close to the basin average value. There was considerable regeneration of phosphorus from the sediments to the water during this storm (Lam and Jaquet 1976) and the constancy of the particulate to total phosphorus ratio indicates considerable regeneration of soluble phosphorus as well as particulate phosphorus from the sediments.

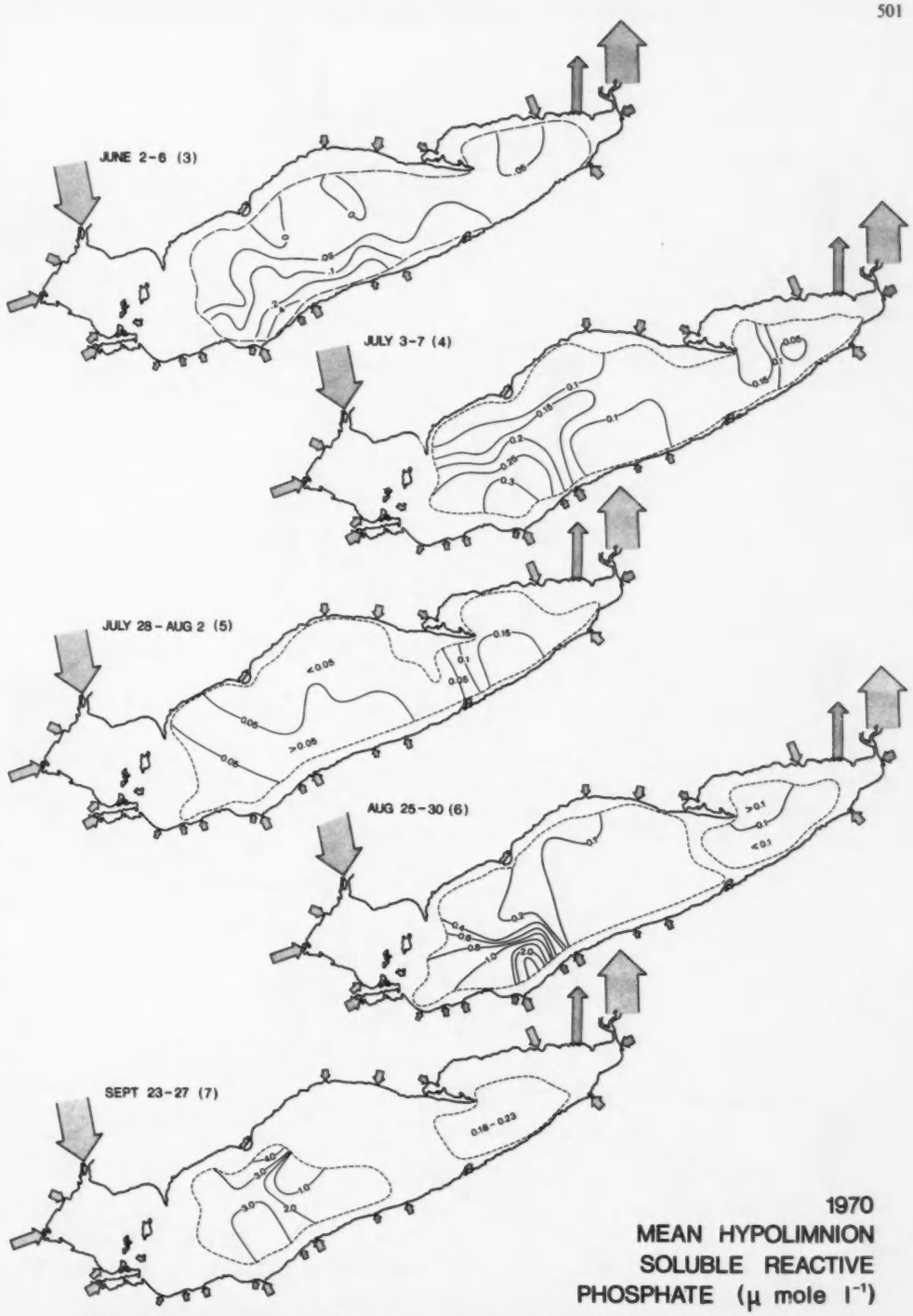


FIG. 12. Mean hypolimnion soluble reactive phosphorus concentrations (June–September 1970).

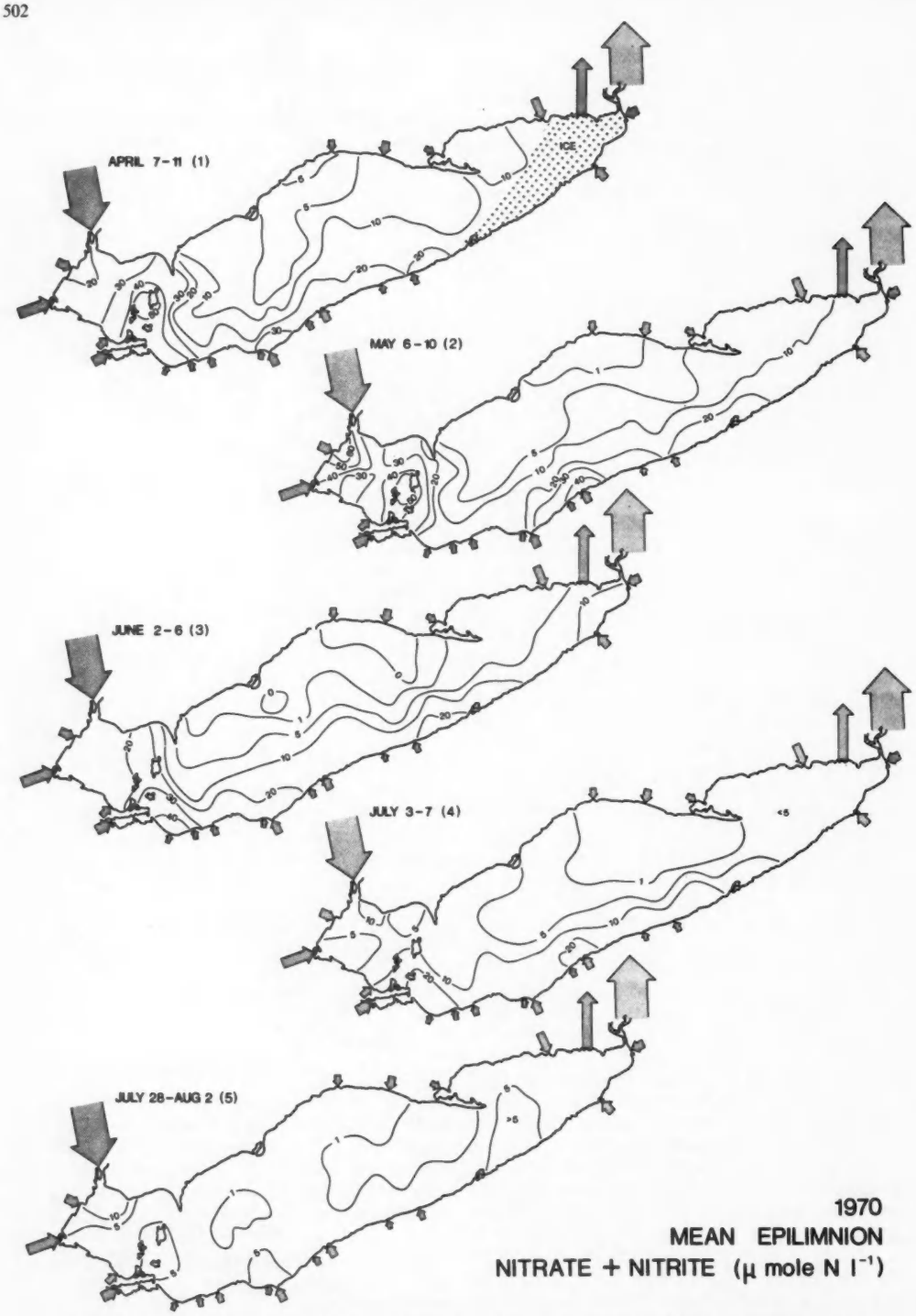


FIG. 13. Lake Erie mean epilimnion nitrate plus nitrite concentrations (April–December 1970).

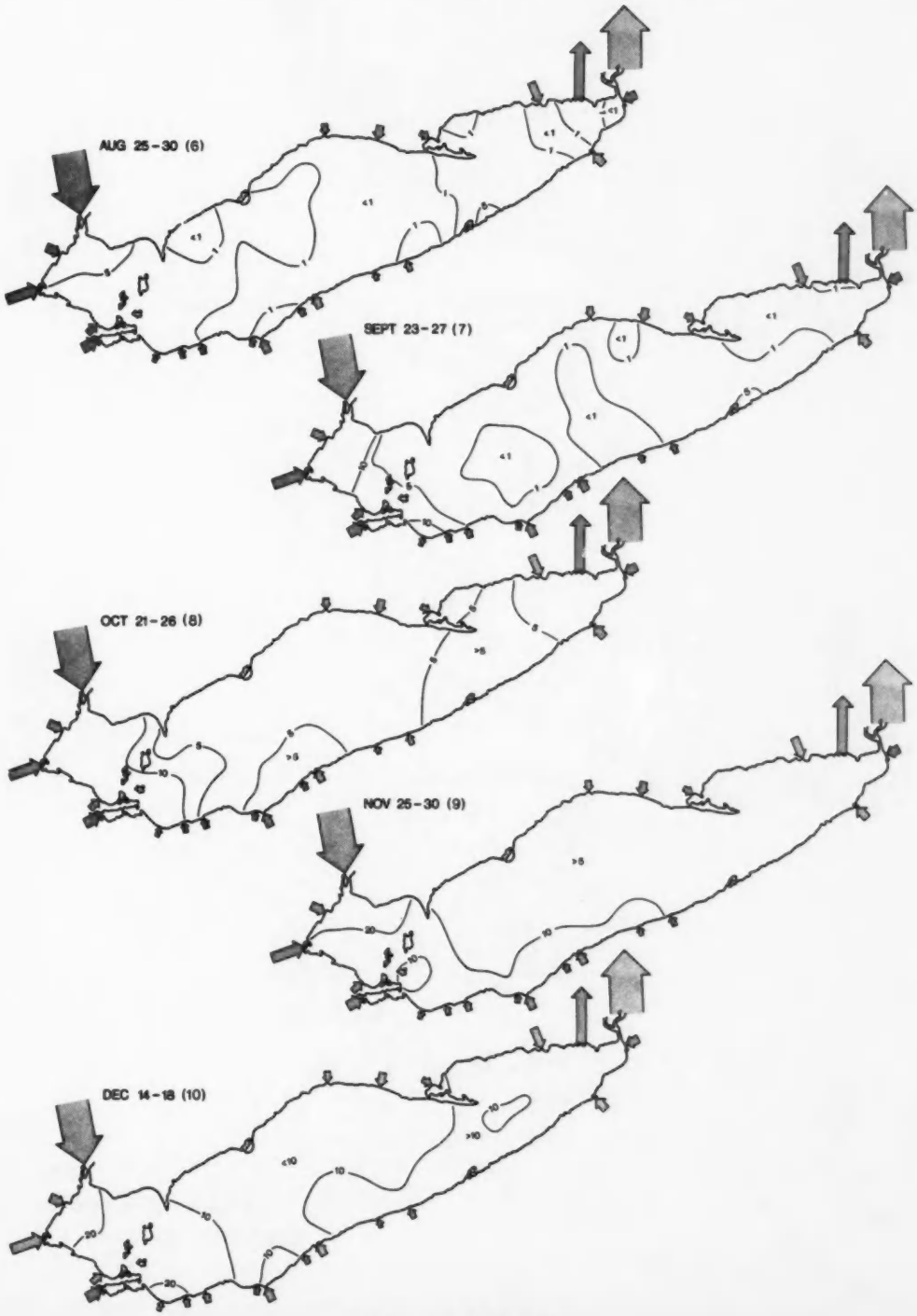


FIG. 13. *(Continued).*

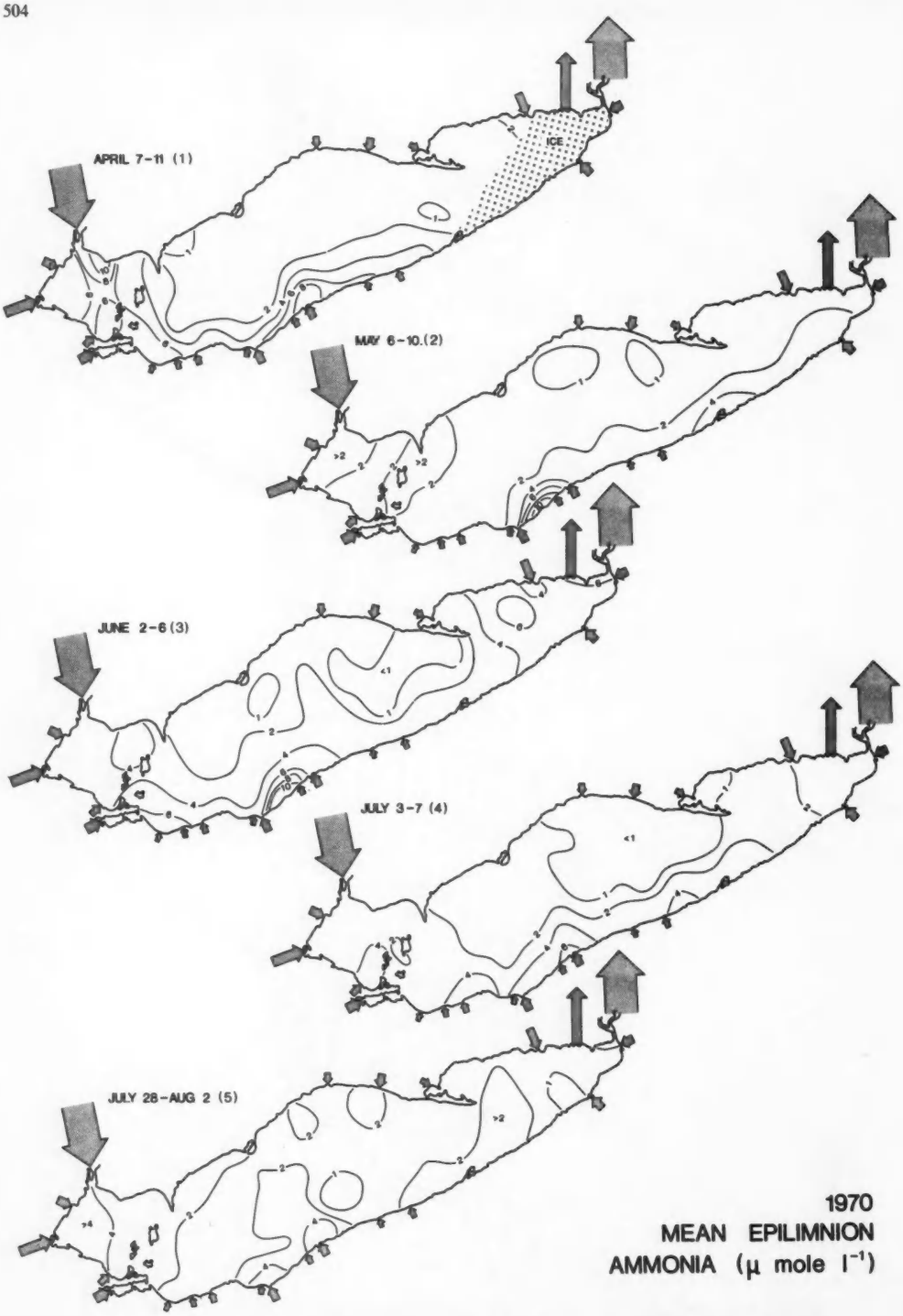


FIG. 14. Lake Erie mean epilimnion ammonia concentrations (April–December 1970).

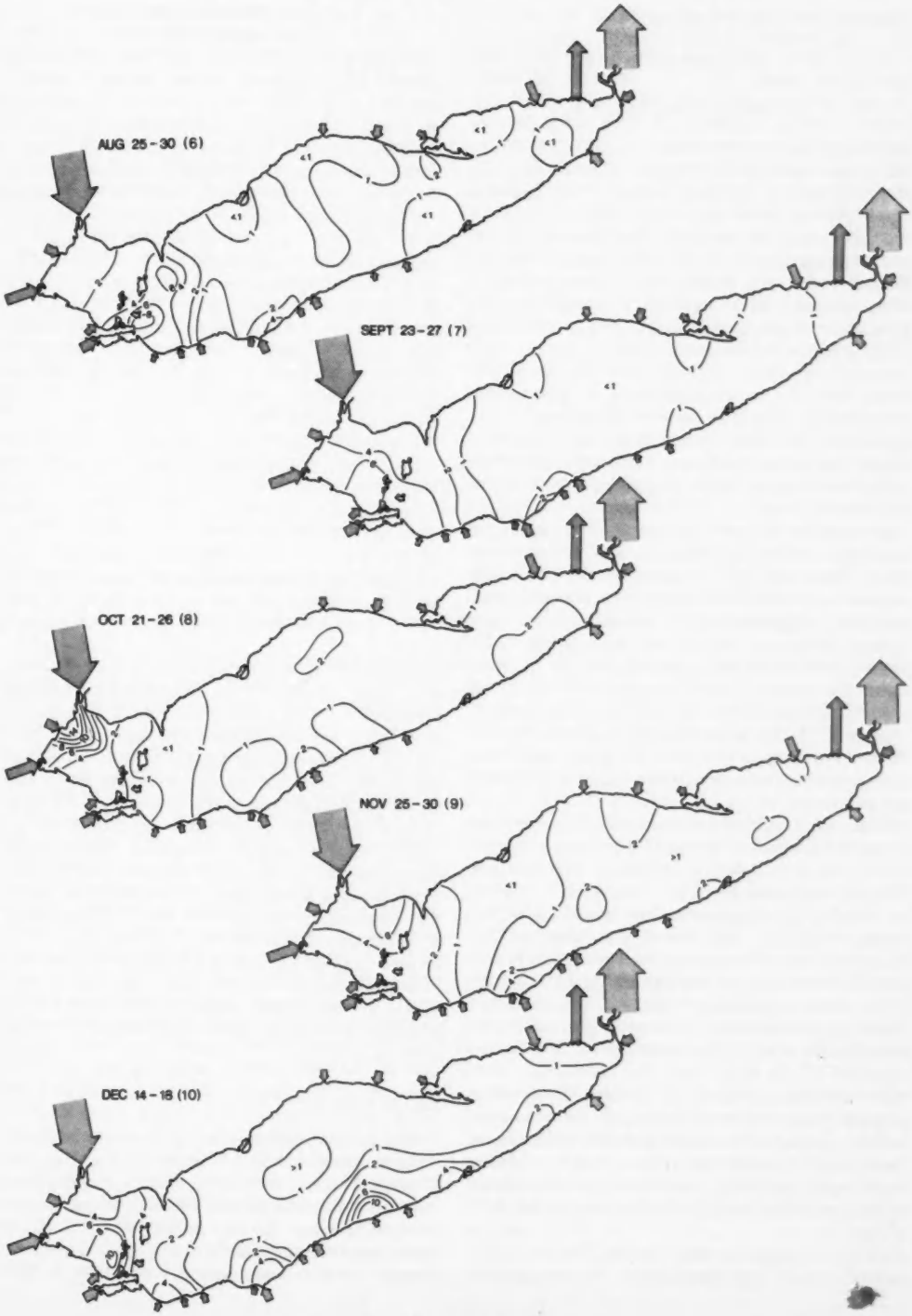


FIG. 14. *(Continued).*

Soluble Inorganic Nitrogen

Nitrate plus nitrite variations across Lake Erie can be quite large (Fig. 13), and it is necessary to look at concentration distribution maps for information on processes affecting these parameters. It would appear from the high concentrations observed in the Western basin and south shore of the Central basin during surveys 1 and 2 that the rivers with their heavy spring flow were the main source of nitrate to the lake. Since there are no large inflows to the northern half of the Central basin, low nitrate plus nitrite concentrations became evident as soon as overall primary production levels increased in the Central basin in May (Glooschenko et al. 1974).

Ammonia shows a low level of variability across the lake in comparison with nitrate plus nitrite (Fig. 14). The variability in mean concentration with the different seasons is also unexpectedly small. However, ammonia does seem to demonstrate the sites of localized inputs to the lake more clearly than nitrate.

Because the observed changes in ammonia concentration are small (Fig. 15) and the decrease in the nitrate plus nitrite concentration during the summer is large, it is possible to consider ammonia as being relatively unimportant in the lake system, with most of the nitrogen required by phytoplankton being absorbed in the nitrate form. However, ammonia is generally the preferred form of nitrogen for phytoplankton uptake (Brezonik 1972; Healy 1973) and is also a product of many organic degradation processes and thus could be expected to show large variations in concentration.

The large nitrate plus nitrite concentration variability and small ammonia concentration variability can be explained in terms of a nitrogen cycle as described by Riley and Chester (1971) in which it is proposed that ammonia is the preferred nitrogen form for algal uptake, but that if the ammonia concentrations decrease below a certain limit, the phytoplankton absorb nitrate rather than ammonia. Further, if the ammonia levels begin to increase because the production of ammonia is greater than that which can be assimilated by the algae, then the nitrification process converts ammonia to nitrate. Thus, it is possible that in each water mass the ammonia would remain at a concentration close to a steady-state concentration for that particular water mass and that the nitrate concentrations would vary considerably. This is seen in the data of Fig. 15.

Thus, it appears that during the summer growth period, the production of the greater

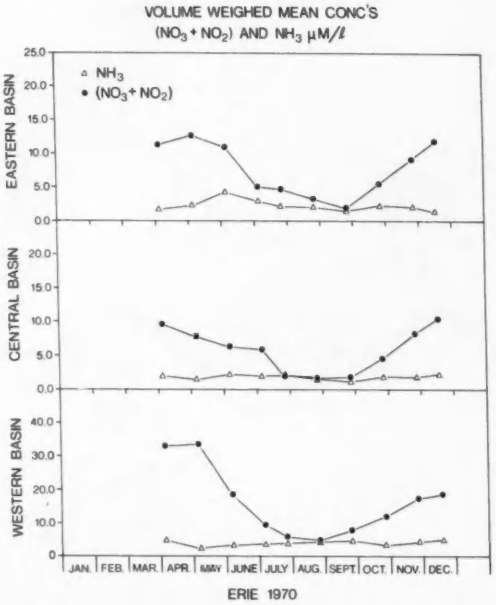


FIG. 15. Mean volume-weighted basin concentrations of ammonia and nitrate plus nitrite in Lake Erie (April–December 1970).

quantities of biomass required more ammonia than was being produced by the nitrogen recycling processes with the result that nitrate was slowly assimilated but not significantly regenerated until the fall. Ammonia concentrations did not change significantly and this nutrient appeared to remain in adequate supply possibly because of nitrogen fixation by blue-green algae (Howard et al. 1970), with subsequent ammonia release upon decomposition of the algal remains. The significant decrease in ammonia concentrations which was observed in the Central and Eastern basins in September coincided with an increase in the diatom population and a disappearance of the blue-green algae. In the late fall, the higher nitrate concentrations were reestablished by increased river inputs and nitrification of excess ammonia.

Soluble Reactive Silica

The general epilimnion distribution of soluble reactive silica (SRS) is shown in Fig. 16. The Western basin is obviously quite different from the rest of the lake in that the SRS concentration fluctuation range is very large. The causes of these large fluctuations are not obvious. Large diatom growths would cause a decrease in SRS

concentrations but strong winds could be expected to increase SRS concentrations by disturbing sediment materials and aiding the dissolution of silica. However, as an example of the unpredictability of the system, low SRS mean concentrations were observed in the Western basin in survey 2 following a period of strong winds and decreased diatom population. It is possible that changes in the SRS concentrations of the Western basin are dominated by changes in SRS concentrations of the Detroit River.

The pattern of the SRS concentration changes in the other two basins is more comprehensible with low levels of SRS in the water coinciding with diatom dominant phytoplankton populations (Munawar and Munawar 1976). During the stratified period the diatom populations diminished and the SRS concentrations slowly built up. This was followed by a concentration decrease in the Central basin at the time of the autumn overturn when the diatom dominance was reestablished. In November and December, the SRS concentrations increased in all parts of the lake in spite of increasing diatom dominance because of the large-scale resupply of SRS from the sediments resulting from complete vertical mixing.

Inshore–Offshore Differences

The stations of the Central basin were divided into three groups: the southshore group of 11 stations with depths less than 18 m, the offshore group of 15 stations with depths greater than 18 m, and the 5 northshore stations with depths less than 18 m. Figures 17–19 demonstrate that there is a striking difference between the nutrient levels in the southshore and offshore waters of the Central basin. They also show that the shallow northern inshore waters are very similar to the deeper offshore waters in their nutrient concentration levels and pattern of concentration change. There are no large sources of external inputs to the lake along the northern shore while there are many along the southern shore. This demonstrates that the high concentration of nutrients along the south shore is not a function of shallow depth and interaction of water with the sediments but is almost entirely due to the external inputs to the lake. Figure 19 shows that the southshore waters have a much higher level of total phosphorus (mean concn. 1.11 μmol liter^{-1}) than the offshore and northshore waters (mean concn. 0.60 μmol liter^{-1}). The mean ratio of the particulate phosphorus to total phosphorus in the southshore waters is 0.62 which is close to the combined offshore–northshore mean ratio of 0.64. This demonstrates that the higher total phosphorus levels in the southshore waters are causing a proportionately higher level of particulate matter in these waters.

Summary and Conclusions

The details of the water sampling procedures and the chemical variables measured during 10 extensive surveys of Lake Erie during 1970 are described here. Distribution maps of the more important variables are presented.

Turbidities in Lake Erie water of greater than 5 J.T.U. are almost entirely due to suspended inorganic material in the water. The observed turbidities in the area between the Western and Central basins make it possible to estimate that there was comparatively little sediment transport between these basins during the year.

The particulate to total phosphorus ratio did not show substantial change between the spring, summer, and fall periods even when large changes in the phytoplankton biomass were observed. The constancy of the particulate to total phophorus ratio was probably the result of the conversion of organic detritus into phytoplankton biomass during growth periods. However, small changes in the ratio when coupled with observations on epilimnion oxygen saturation were a good indicator of the major periods of phytoplankton growth. Further, no significant change in the particulate to total phosphorus ratio was observed during a period of phosphorus release resulting from sediment disturbance. This indicated that about 40% of the phosphorus released from the sediments was in the soluble form.

It is quite obvious that the anoxic conditions in the Central basin hypolimnion were unsuitable for fish life and that they resulted in the regeneration of large amounts of phosphorus from the sediments. Generally, the oxygen conditions in the Eastern basin were adequate for fish life.

The large erosional inputs of apatite phosphorus to the Central basin are not manifested in the lake by high phosphorus concentrations in the waters close to the sources of input. It is considered that these erosional inputs of phosphorus do not play a significant part in the phosphorus cycle of the lake.

Nitrate plus nitrite levels in the lake showed a large decrease during the summer while the ammonia levels remained almost constant. However, ammonia was probably the preferred form of nitrogen for algal uptake; these apparently inconsistent results can be explained in terms of the nitrogen cycle.

The northshore and offshore concentrations of soluble nitrogen and phosphorus in the Central

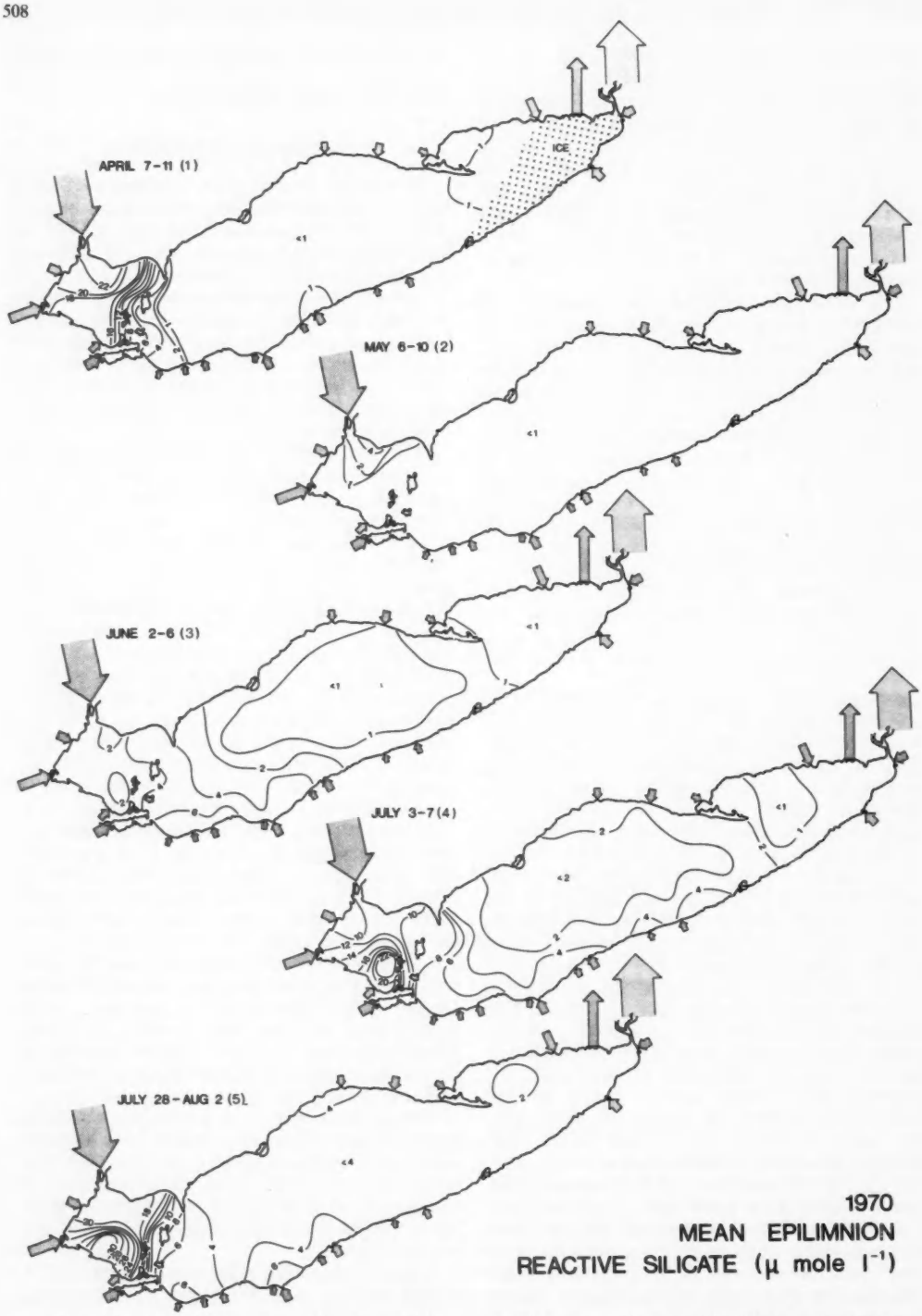
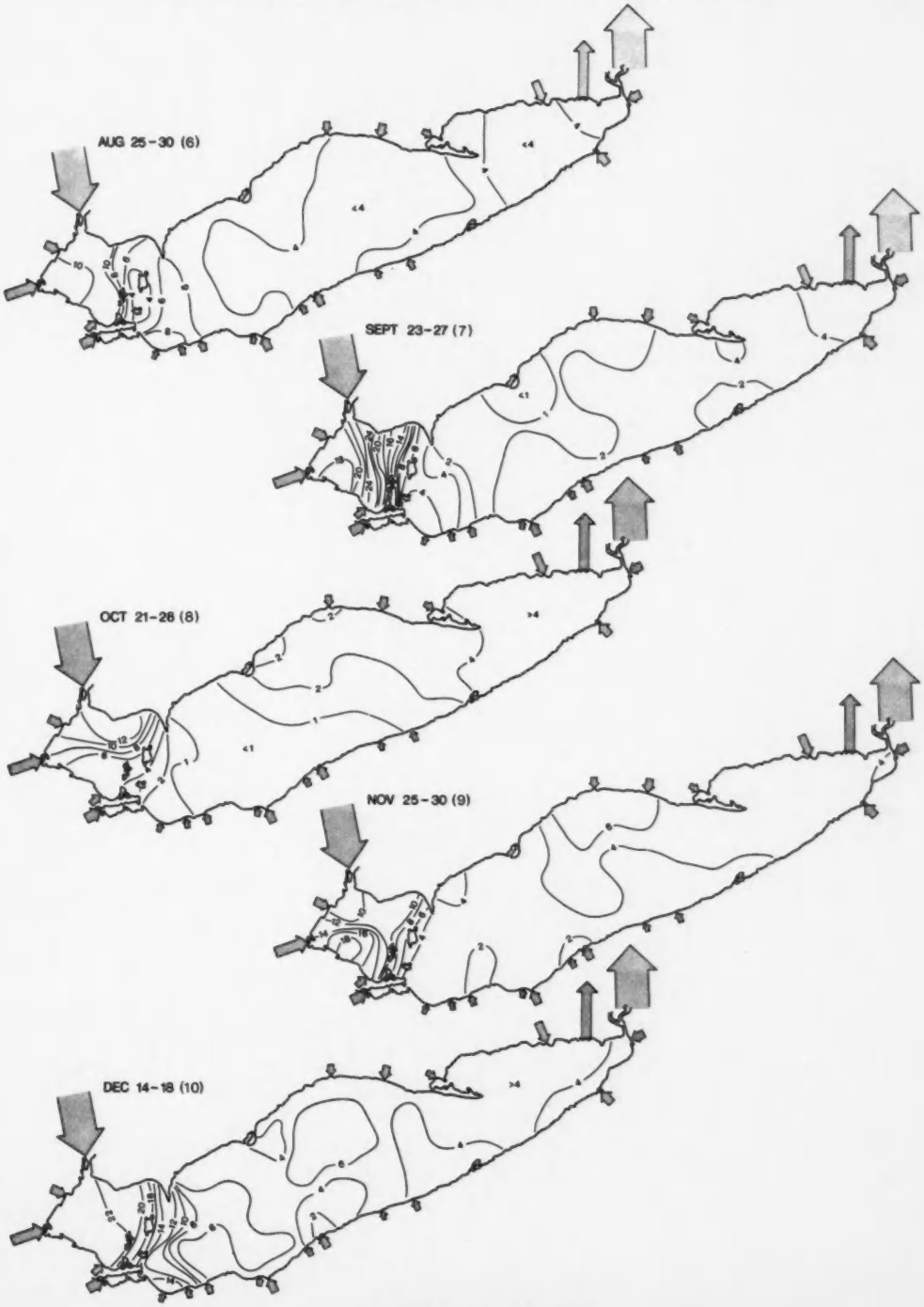


FIG. 16. Lake Erie mean epilimnion soluble reactive silica concentrations (April–December 1970).



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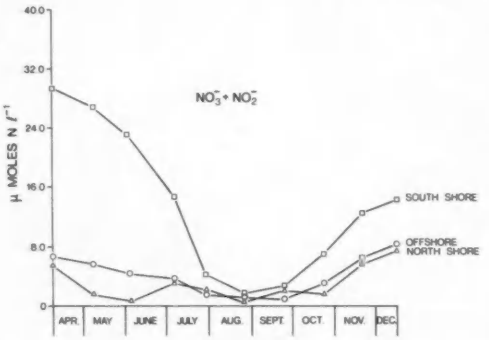

FIG. 17. Concentrations of $NO_3^- + NO_2^-$ in the Central basin of Lake Erie, 1970.

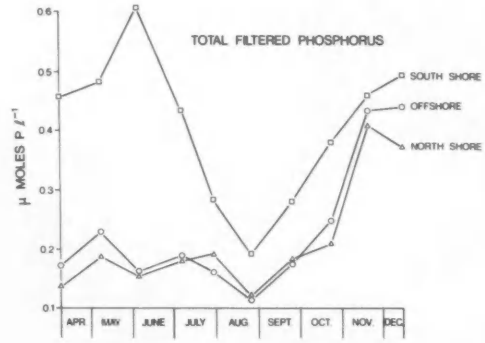

FIG. 18. Concentrations of total filtered phosphorus in the Central basin of Lake Erie, 1970.

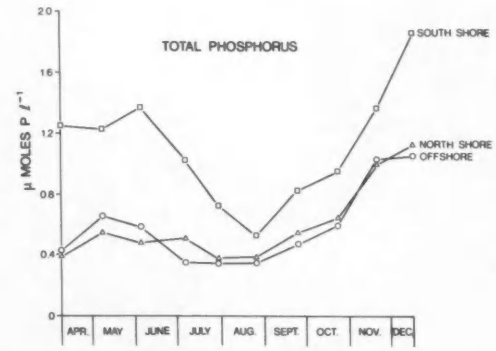

FIG. 19. Concentrations of total phosphorus in the Central basin of Lake Erie, 1970.

basin were very similar but much lower than the southshore concentrations. This indicated that the high southshore concentrations were a result of shoreline inputs and not a result of the release of soluble nutrients from sediments disturbed by wave action.

Acknowledgments

I thank the Technical Operations Section and the Water Quality Division at the Canada Centre for Inland Waters for performing the huge task of collecting and analysing the samples which form the data base for this study. In particular, I thank C. H. Chan who did the time-consuming work of careful data processing.

BREZONIK, P. O. 1972. Nitrogen, sources and transportation in natural waters, p. 1–50. *In* H. E. Allen and J. R. Kramer [ed.] Nutrients in natural waters. John Wiley and Sons, Inc., New York, N.Y.

BURNS, N. M. 1976a. Oxygen depletion in the Central and Eastern basins of Lake Erie during 1970. J. Fish Res. Board Can. 33: 512–519.

1976b. Nutrient budgets in Lake Erie. J. Fish. Res. Board Can. 33: 520–536.

BURNS, N. M., F. ROSA, AND C. H. CHAN. 1976a. Lake Erie water chemistry data 1970–71. Can. Cent. Inland Waters, Burlington, Ont. Pap. 16: (In press)

BURNS, N. M., AND C. ROSS. 1972. Oxygen–nutrient relationships within the Central Basin of Lake Erie, p. 85–119. *In* N. M. Burns and C. Ross [ed.] "Project Hypo." Can. Cent. Inland Waters, Pap. 6: 180 p.

BURNS, N. M., J. D. H. WILLIAMS, A. L. W. KEMP, J.-M. JAQUET, AND D. C. L. LAM. 1976b. A phosphorus budget for Lake Erie. J. Fish. Res. Board Can. 33: 564–573.

GÄCHTER, R., R. A. VOLLENWEIDER, AND W. A. GLOOSCHENKO. 1974. Seasonal variations of temperature and nutrients in surface waters of Lakes Ontario and Erie. J. Fish. Res. Board Can. 31: 275–290.

GLOOSCHENKO, W. A., J. E. MOORE, AND R. A. VOLLENWEIDER. 1974. Spatial and temporal distribution of chlorophyll *a* and pheopigments in surface waters of Lake Erie. J. Fish. Res. Board Can. 31: 265–274.

HARTLEY, R. P. 1968. Bottom currents in Lake Erie. Proc. Conf. Great Lakes Res. 11: 398–405.

HEALEY, F. P. 1973. Inorganic nutrient uptake and deficiency in algae. C.R.C. Crit. Rev. Microbiology. p. 69–113.

HOWARD, D. L., J. I. FREW, P. M. PFISTER, AND P. R. DUGAN. 1970. Nitrogen fixation in Lake Erie. Science 169: 61–72.

KEMP, A. L. W., R. L. THOMAS, C. I. DELL, AND J.-M. JAQUET. 1976. Cultural impact on the geochemistry of sediments in Lake Erie. J. Fish. Res. Board Can. 33: 440–462.

KRAMER, J. R. 1961. The Chemistry of Lake Erie. Proc. Conf. Great Lakes Res. 4: 27–56.

KRAMER, J. R., H. E. ALLEN, G. W. BAULNE, AND N. M. BURNS. 1970. Lake Erie time study. Can. Cent. Inland Waters. Pap. 6: 50 p.

LAM, D. C. L., AND J.-M. JAQUET. 1976. Computations of physical transport and regeneration of phosphorus in Lake Erie. J. Fish. Res. Board Can. 33: 550–563.

LEACH, J. H. 1975. Seston composition in the Pt. Pelee area of Lake Erie. Int. Rev. Gesamten Hydrobiol. 60. 347–357.

MUNAWAR, M., AND I. M. F. MUNAWAR. 1976. Lakewide study of phytoplankton biomass and its species composition in Lake Erie, April–December 1970. J. Fish. Res. Board Can. 33: 581–600.

PHILBERT, F. J., AND W. J. TRAVERSY. 1973. Methods of sample treatment and analysis of lake waters and precipitation samples. Proc. Conf. Great Lakes Res. 16: 294–310.

RILEY, J. P., AND R. CHESTER. 1971. Introduction to marine chemistry. Academic Press, New York, N.Y.

SIMONS, T. J. 1976. Continuous dynamical calculations of water transports in Lake Erie in 1970. J. Fish. Res. Board Can. 33: 371–384.

STRACHAN, W. M. J. 1973. A statistical examination of Great Lakes monitor data at the Canada Centre for Inland Waters. Proc. Conf. Great lakes Res. 16: 949–957.

WEILER, R. R., AND V. K. CHAWLA. 1968. The chemical composition of Lake Erie. Proc. Conf. Great Lakes Res. 11: 593–608.

Oxygen Depletion in the Central and Eastern Basins of Lake Erie, 1970

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BURNS, N. M. 1976. Oxygen depletion in the Central and Eastern basins of Lake Erie, 1970. J. Fish. Res. Board Can. 33: 512–519.

The hypolimnetic oxygen depletion rate is required to be measured to an accuracy of approximately 3% if major changes in the trophic state of Lake Erie are to be documented within about 5 yr of the change commencing. Depletion rate measurements of this level of accuracy require a knowledge of the reoxygenation of the Central and Eastern basin hypolimnia and the transport of oxygen between the hypolimnia. A model is described here that enables an estimate of the degree of hypolimnetic reoxygenation to be made. The transport of oxygen between the hypolimnia was estimated by means of a hydrodynamic model. The resultant calculations gave areal hypolimnetic depletion rate values of 0.43 and 0.87 g O_2 m^{-2} day^{-1} for the Central and Eastern basins, respectively, and volumetric depletion rates of 0.13 and 0.057 g O_2 m^{-3} day^{-1}, respectively, for these basins during 1970.

BURNS, N. M. 1976. Oxygen depletion in the Central and Eastern basins of Lake Erie, 1970. J. Fish. Res. Board Can. 33: 512–519.

On doit mesurer le taux de déplétion de l'oxygène hypolimnétique avec une précision d'environ 3% si l'on veut documenter les changements majeurs dans l'état trophique du lac Erié en dedans de 5 a environ après le début du changement. Les mesures des taux de déplétion à ce niveau de précision requièrent une connaissance de la réoxygénation de l'hypolimnion des bassins central et oriental, et du transport de l'oxygène d'un hypolimnion à l'autre. Nous décrivons un modèle permettant d'estimer le degré de réoxygénation hypolimnétique. Un modèle hydrodynamique a permis d'estimer le transport d'oxygène entre les hypolimnions. Les calculs qui en résultent donnent, pour les taux de déplétion hypolimnétique par superficie, des valeurs de 0.43 et 0.87 g $O_2 \cdot m^{-2} \cdot jour^{-1}$ pour les bassins central et oriental respectivement, et des taux de déplétion en volume de 0.13 et 0.057 g $O_2 \cdot m^{-3} \cdot jour^{-1}$ respectivement pour ces bassins en 1970.

Accepted November 18, 1975 Accepté le 18 novembre 1975

THE degradation of the water quality of Lake Erie since the turn of the century is well known. One of the most quantitative measures of this degradation has been the increase in the oxygen depletion rate of the Central basin hypolimnion waters. Dobson and Gilbertson (1972) have estimated that this depletion rate has increased from 0.05 g O_2 m^{-3} day^{-1} in 1929 to 0.11 g O_2 m^{-3} day^{-1} in 1970 at a rate of 0.0025 g O_2 m^{-3} day^{-1} yr^{-1}; indicating an increase in the rate of about 3.0% per year during the past 20 years.

As a result of the Great Lakes Water Quality Agreement (U.S. Department of State 1972), plans have been drawn up to improve the water quality of Lake Erie by reducing the loading of phosphorus and other nutrients to the lake.

If it is appropriate to use a future decrease in the Central basin hypolimnetic oxygen depletion rate as an indicator of the improvement of the lake, then a sensitive and accurate method of measuring this value is required. If the mean annual hypolimnetic oxygen depletion rate decreased at a rate similar to the past rate of increase, the depletion rate decrease would have to be measured to an accuracy of ±3% for about 5 consecutive years before the decrease in the oxygen depletion rate could be confidently assigned to the improved water quality of the lake and not considered to be a result of annual variations in weather.

One of the main factors preventing the accurate determination of the mean hypolimnion oxygen depletion rate is the highly variable degree of reoxygenation of the hypolimnion waters. Increases in the Central basin hypolimnion volume of up to 25% in 10 days have been observed (Burns and Ross 1972a) as a result of the incorporation of oxygenated mesolimnion and epilimnion water into the hypolimnion.

A method of calculating the degree of reoxygenation of a hypolimnion in the interval between observational surveys is presented here. The basic approach has been to regard heat as a

Printed in Canada (J4067)
Imprimé au Canada (J4067)

conservative quantity within the hypolimnion and to use the hypolimnetic heat budget for the estimation of the degree of interchange between the mesolimnion and hypolimnion water masses. A knowledge of the degree of this water mass interchange has then made it possible to estimate the degree of hypolimnetic reoxygenation. This method of estimation has been used in the calculation of values for the hypolimnetic oxygen depletion rate in both the Central and Eastern basins from the data of five of the 1970 ship surveys of the M.V. *Martin Karlsen* (Burns 1976a).

The Mesolimnion Exchange Model

The Mesolimnion Exchange Model has been developed so that the transport of materials between the hypolimnion and epilimnion during the stratified season could be estimated. These estimates make it possible to calculate hypolimnion oxygen budgets and epilimnion nutrient budgets.

There are a number of processes occurring in the interaction of a hypolimnion with a mesolimnion, the most important of these being:

1) The incorporation of a layer of the mesolimnion downwards into the hypolimnion leading to a hypolimnion volume increase;

2) The incorporation of a layer of the hypolimnion upwards into the mesolimnion leading to a hypolimnion volume decrease;

3) Downwelling of a part of the mesolimnion and epilimnion into the hypolimnion with the simultaneous upwelling of a part of the hypolimnion elsewhere into the mesolimnion or epilimnion. This process would lead to an exchange of water between hypolimnion and epilimnion with no volume change in the hypolimnion.

These processes can happen simultaneously but in this analysis it will be assumed that only one process occurs at a time. Further, if in process (3) the downwelling is greater than the simultaneous upwelling, then process (1) would account for that portion of the exchange which manifests itself as increased hypolimnion volume. Similarly, process (2) would be used in the calculation of an exchange in which the downwelling was less than the upwelling.

Calculation of the Incorporated Heat

The effective heat budget equation for the difference in hypolimnion heat content between the two surveys, no. 1 and no. 2, would be:

$$H_2 - H_1 = H_i + H_e + H_c - H_s + H_{ad} \qquad (1)$$

where, H_2 = Heat content observed during survey no. 2 = V_2T_2;
H_1 = Heat content observed during survey no. 1 = V_1T_1;
V_2 = Hypolimnion volume survey no. 2, T_2 = mean volume-weighted hypolimnion temperature survey no. 2.
V_1 = Hypolimnion volume survey no. 1, T_1 = mean volume-weighted hypolimnion temperature survey no. 1.
H_i = Heat incorporated into the hypolimnion from the mesolimnion.
= ΔVT_d or ΔVT_h
ΔV = volume of water incorporated
= $V_2 - V_1 - V_{ad}$
H_e = Heat transferred into the hypolimnion by the exchange process with no associated hypolimnion volume change.
H_c = Heat conducted through the thermocline.
H_s = Heat conducted into the sediment.
H_{ad} = Heat advected into the hypolimnion horizontally from a contiguous hypolimnion.
= $V_{ad}T_{ad}$
V_{ad} = Volume added to the hypolimnion from a contiguous hypolimnion.
T_{ad} = Mean temperature of water added from a contiguous hypolimnion.

The mean temperature of the water incorporated downwards from the mesolimnion, T_d would be given by

$$T_d = T_h + \left(\frac{\Delta V}{A_h} \times \frac{\Delta T_g}{2}\right) \qquad (2)$$

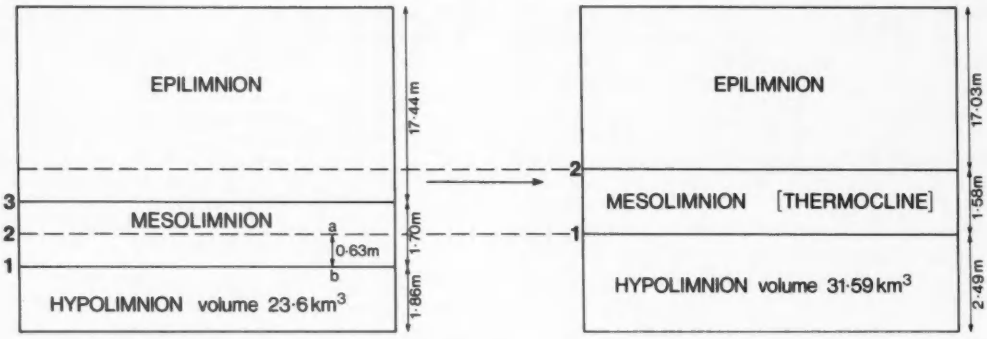
where, T_h = mean of both of the mean area-weighted temperatures for the top of the hypolimnion, surveys no. 1, no. 2.
A_h = mean area of the hypolimnion
ΔT_g = mean area-weighted mesolimnion temperature gradient

The temperature of the water moving downwards is dependent upon the area of downwelling for a given volume of incorporation because the smaller the area over which a certain volume is to be incorporated, the greater will be the thickness and mean temperature of the incorporated layer, as demonstrated by the $(\Delta V/A_h)$ term. The heat incorporated into the hypolimnion, H_i, would be given by ΔVT_d.

If instead of a volume increase, the volume of the hypolimnion decreased, then ΔV would represent the volume of the water incorporated upwards into the mesolimnion and the heat connected with this incorporation upwards would be given by the value of ΔVT_h.

The Difference Between the Erosion and Exchange Models

The Mesolimnion Erosion Model described in Project Hypo (Burns and Ross 1972a) assumed



Heat transferred to hypolimnion during period from Survey 2 to Survey 4 i.e. $[H_i]\ 2-4 = 12{\cdot}09° \times 7{\cdot}99\ km^3$
$= 0{\cdot}966 \times 10^{14}$ kilocalories

FIG. 1. A schematic representation of the Mesolimnion Erosion Model showing heat and volume transfers which occurred between surveys 2 and 4.

that only processes (1) and (2) described above, occurred during the study and that only one of these processes occurred during any one survey interval of 4 days. In this early model, a further assumption was made that the upward or downward incorporation occurred across the whole basin instead of across only a part of the basin. Equation (2) was used for the calculation of the mean temperature of the water incorporated downward and this resulted in the lowest possible mean temperature being calculated for the incorporated water because of the maximum value used for A_h. An example of how the Mesolimnion Erosion Model was used is shown in Fig. 1. During the complete study, the mean hypolimnion temperature increased by 1.83 C and the simple Mesolimnion Erosion Model was able to account for 91% of the temperature rise, i.e. 1.62 C. This led to the conclusion that the major process was basin-wide incorporation of mesolimnion water into the hypolimnion in the case of a hypolimnion volume increase. The basin-wide incorporation of water into the mesolimnion would also apply in the case of a hypolimnion volume decrease.

Although the Mesolimnion Erosion Model was nearly adequate in representing the incorporation phenomena, it did not take process (3) into account, which process can add significant quantities of heat to the hypolimnion without a volume change. It is this inadequacy which probably led to the temperature prediction by the Mesolimnion Erosion Model being low. A further inadequacy of this model is that the heat transfer calculations require accurately measured hypolimnion volumes which are difficult measures to achieve.

The Mesolimnion Exchange Model is essentially the same as the Erosion Model except that a term for the exchange heat, H_e, has been added. By means of this new model, observed increases in the mean hypolimnion temperature can be accounted for. This updated model is based primarily on volume-weighted and area-weighted hypolimnion and mesolimnion temperatures.

Equation (1) can now be rewritten:

$$H_s = H_2 - H_1 - H_t - H_{sd} - H_c + H_e \qquad (3)$$

and, since all the terms on the right-hand side of equation (3) can be estimated, a value for H_e can be obtained. The calculation of H_c and H_s,

which are comparatively small terms and which almost cancel each other, is described by Burns and Ross (1972a).

Use of Exchange Heat in Other Exchange Calculations

Once an estimate of the heat transferred between the mesolimnion and hypolimnion by the exchange processes has been obtained, this exchange heat quantity can be used to estimate the quantity of materials transferred between the mesolimnion and hypolimnion by the same processes.

If, V_e = volume of water exchanged between hypolimnion and mesolimnion between surveys not resulting in hypolimnion volume change
H_{ed} = heat added downwards during the exchange process
$= V_eT_d$
H_{eu} = heat added upwards during the exchange process
$= V_eT_h$
A = area of downwelling

$$\text{then, } H_e = H_{ed} - H_{eu}$$
$$= V_e\left(T_h + \left(\frac{V_e}{A} \times \frac{\Delta T_g}{2}\right)\right) - V_eT_h$$
$$= V_e^2\left(\frac{\Delta T_g}{2A}\right) \quad (4)$$

It can be seen that the volume exchanged depends on the area of the downwelling. In this instance, the heat budget is only of interest in that it yields information concerning the hypolimnion budgets of nonconservative quantities, (e.g. oxygen) and as such the knowledge of the downwelling area is not necessary as shown below in equation (5). If the oxygen mixed down by water undergoing the exchange process is considered to be mixed down in the same manner as the exchange heat then the quantity of oxygen exchanged is given by O_e,

$$\text{where, } O_e = \Delta O_g\left(\frac{V_e^2}{2A}\right)$$
$$= \Delta O_g\left(\frac{H_e}{\Delta T_g}\right) \quad (5)$$

if, ΔO_g = mean area-weighted oxygen concentration gradient through the mesolimnion

Thus, the information on the heat budget and on the mesolimnion gradient of oxygen concentration (or any other species) permits an estimate of the quantity of oxygen or other material which has been exchanged between the mesolimnion and hypolimnion by the exchange mechanism in the interval between ship surveys. However, information is also required on the mean temperature gradients of the mesolimnion for both surveys. In the case of soluble nutrients, the hypolimnion concentration of the nutrients is often greater than the epilimnion concentration, and the Exchange Model can then be used to estimate the transport of soluble nutrients to the mesolimnion and/or the epilimnion by the exchange mechanism.

It is also necessary to calculate the transfer of oxygen or other material to or from the hypolimnion by the incorporation mechanism.

The oxygen incorporated into the hypolimnion in the case of a hypolimnion volume increase is given by ΔVO_d, where

$$O_d = O_h + \left(\frac{\Delta V}{A_h} \times \frac{\Delta O_g}{2}\right) \quad (6)$$

O_d = mean concentration of oxygen in water incorporated downwards into the hypolimnion
O_h = mean area weighted hypolimnion concentration of oxygen

The oxygen incorporated upwards into the mesolimnion in the case of a hypolimnion decrease is given by ΔVO_h.

The quantity of oxygen depleted is given by

$$\Delta O = V_1O_1 + O_e + O_i + V_{ad}O_{ad} - V_2O_2 \quad (7)$$

where, ΔO = quantity of oxygen which has been depleted from the hypolimnion
O_1 = volume weighted mean hypolimnion oxygen concentration, Survey no. 1
O_2 = volume weighted mean hypolimnion oxygen concentration, Survey no. 2
O_i = oxygen incorporated upwards or downwards from the hypolimnion
$= \Delta VO_d$ or ΔVO_h (whichever is appropriate)
O_{ad} = mean oxygen concentration of water advected from a contiguous hypolimnion.

Methods

The methods used for the calculation of the volume-weighted mean hypolimnion temperature and oxygen concentrations and the hypolimnetic quantities of heat and oxygen in the Central and Eastern basin hypolimnia have been described by Burns (1976b).

The oxygen depletion rate calculation for the Central basin between surveys 4 and 5 is shown in detail using the method and variables defined for the Mesolimnion Exchange Model above. In this case, there was a flow of 15.52 km^3 of water from the Eastern basin hypolimnion to the Central basin. Survey 4 variables carry the subscript 1 and survey 5 variables carry the subscript 2; O_x is the abbreviation for oxygen.

TABLE 1. Oxygen depletion in the Central basin hypolimnion, 1970.

Survey interval	Mean hypo thickness (m)	Hypolimnia interaction: Volumetric depletion rate ($g\ m^{-3}\ day^{-1}$)	Hypolimnia interaction: Areal depletion rate ($g\ m^{-2}\ day^{-1}$)	No hypolimnia interaction: Volumetric depletion rate ($g\ m^{-3}\ day^{-1}$)	No hypolimnia interaction: Areal depletion rate ($g\ m^{-2}\ day^{-1}$)	Means: Volumetric ($g\ m^{-3}\ day^{-1}$)	Means: Areal ($g\ m^{-2}\ day^{-1}$)
3–4	3.4	0.13	0.44	0.11	0.39	0.12	0.42
4–5	3.5	0.18	0.62	0.11	0.38	0.15	0.50
5–6	3.1	0.13	0.40	0.12	0.36	0.13	0.38
6–7	(2.7)	(0.08)	(0.23)	(0.08)	(0.23)		
Mean	3.3	0.15	0.49	0.11	0.38	0.13	0.43

TABLE 2. Oxygen depletion in the Eastern basin hypolimnion, 1970.

Survey interval	Mean hypo thickness (m)	Hypolimnia interaction: Volumetric depletion rate ($g\ m^{-3}\ day^{-1}$)	Hypolimnia interaction: Areal depletion rate ($g\ m^{-2}\ day^{-1}$)	No hypolimnia interaction: Volumetric depletion rate ($g\ m^{-3}\ day^{-1}$)	No hypolimnia interaction: Areal depletion rate ($g\ m^{-2}\ day^{-1}$)	Means: Volumetric ($g\ m^{-3}\ day^{-1}$)	Means: Areal ($g\ m^{-2}\ day^{-1}$)
3–4	18.8	0.083	1.55	0.082	1.54	0.083	1.55
4–5	14.2	0.055	0.78	0.043	0.62	0.049	0.70
5–6	13.1	0.054	0.71	0.052	0.68	0.053	0.70
6–7	12.8	0.042	0.54	0.042	0.54	0.042	0.54
Mean	14.7	0.059	0.90	0.055	0.84	0.057	0.87

$H_1 = 354.4 \times 10^{12}$ kcal
$H_2 = 316.1 \times 10^{12}$ kcal
$V_1 = 40.02\ km^3$; $O_1 = 265.1 \times 10^6$ kg O_x
$V_2 = 30.47\ km^3$; $O_2 = 122.2 \times 10^6$ kg O_x
$V_{ad} = 15.52\ km^3$ (from Eastern to Central basin)
$\Delta V = -25.07\ km^3$
$T_{ad} = 8.24$ C; $O_{ad} = 8.89$ g $O_x m^{-3}$
$T_h = 10.26$ C; $O_h = 5.46$ g $O_x m^{-3}$
$H_c - H_s = 0.0$

Thus, $H_e = (316.1 - 354.4 + 257.2 - 127.9 - 0.0) \times 10^{12}$ kcal
$= 91.0 \times 10^{12}$ kcal
$= 91.0 \times 10^9$ C·m^3
$\Delta T_g = 5.52$ C m^{-1}
$\Delta O_g = 2.02$ g $O_x m^{-4}$

Therefore, $O_e = \frac{2.02\ g}{5.52\ C\ m^{-1}} O_x m^{-4} \times 91.0 \times 10^9$ C·m^3
$= 33.3 \times 10^6$ kg O_x

Thus, $\Delta O = (265.1 + 33.3 - 137.0 + 138.0 - 122.2) \times 10^6$ kg O_x
$= 177.2 \times 10^6$ kg O_x

Areal oxygen depletion rate ($A_h = 10.2 \times 10^9 m^2$)

$$= \frac{177.2 \times 10^9\ g}{10.2 \times 10^9\ m^2} \times \frac{1}{28\ days}$$
$$= 0.62\ g\ m^{-2}\ day^{-1}$$

Volume oxygen depletion rate ($V_{av} = 35.24\ km^3$)

$$= \frac{177.2 \times 10^9\ g}{35.24 \times 10^9\ m^3} \times \frac{1}{28\ days}$$
$$= 0.18\ g\ m^{-3}\ day^{-1}$$

Tables 1 and 2 give the oxygen depletion rates calculated in two slightly different ways. In the first method, the flows between the Central and Eastern basin hypolimnion as estimated by Simons (Simons 1976) have been accepted as correct and used in the calculation. In the second set of calculations, the assumption was made that there was no flow between the hypolimnia. Simons considered his flow estimates as probably being greater than the real flow, and thus the values given here are the extremes of the range that contains the correct value. In the subsequent calculations and discussion, the oxygen depletion rates used are those which are the simple mean of the two rates calculated with and without hypolimnia exchange.

Results

The distribution maps showing the observed oxygen concentrations on which this analysis is based, have been published by Burns (1976a). The complete data set is available on request (Burns et al. 1976). The areal and volumetric hypolimnetic oxygen depletion rates for the Central and Eastern basins for each survey interval have been calculated and are shown in Tables 1 and 2.

In Project Hypo (Burns and Ross 1972a), an accurate estimate of the oxygen depletion rate for the Central basin during the month of August 1970 is given as 0.39 g O_2 m^{-2} day^{-1}. The value given in Table 1 for the oxygen depletion

rate calculated from the different data sets collected during the two August surveys of the M.V. *Martin Karlsen*, survey 5 (July 28 to Aug. 2) to survey 6 (Aug. 25–30) and the Mesolimnion Exchange Model is 0.38 gm O_2 m^{-2} day. This good agreement gives increased confidence in the model. The oxygen depletion rates calculated for the Central basin between surveys 6 and 7 are not included in the mean. During this period, much of the hypolimnion was anoxic (Burns 1976a) and the lowered oxygen depletion rate is a result of the unavailability of oxygen.

Discussion

Is can be seen from the detailed calculation shown above that the oxygen added to the hypolimnion by the exchange process comprised about 19% of the oxygen depleted from the hypolimnion during the period between surveys 4 and 5. The oxygen added to the Central basin hypolimnion by the exchange process during 1970 was in the order of 12% of the oxygen depleted, while the oxygen added by this process to the Eastern basin hypolimnion, constituted approximately 5% of the oxygen depleted from that water mass. It is obvious then that oxygen transfer across the mesolimnion by exchange must be taken into account in Lake Erie if oxygen depletion calculations of an accuracy of ± 3% are to be achieved.

The mean difference in the rates (Tables 1 and 2) calculated with and without hypolimnia interaction was about 8% in the Eastern basin and 25% in the Central basin. This demonstrates that the oxygen advected between the hypolimnia of the two basins must be carefully measured if the uncertainty in the depletion rate resulting from advection of hypolimnetic waters is to be held within 3% of the total amount of oxygen depleted.

A knowledge of both the areal and volumetric hypolimnetic depletion rates is necessary in order to characterize the oxygen consuming properties of a hypolimnion. The data in Tables 1 and 2 show that the Central basin mean volumetric hypolimnion oxygen depletion rate is approximately twice that of the Eastern basin rate while the areal depletion rate is approximately half that of the Eastern basin rate.

The mean particulate phosphorus content of the Central basin hypolimnion for the summer season was 9.6 mg P m^{-3} and that of the Eastern basin was 4.7 mg P m^{-3}. The weight ratio of particulate carbon to particulate phosphorus in the material settling into the Central basin hypolimnion in August 1970, was observed to be 47:1 (Burns and Ross 1972a). If this ratio is applied to both basins, it would mean that the particulate organic carbon concentration was in the order of 0.45 g C m^{-3} in the Central basin hypolimnion and 0.22 g C m^{-3} in the Eastern basin hypolimnion. It is possible that the volumetric oxygen depletion rate of the Central basin was twice that of the Eastern basin because the hypolimnetic particulate organic carbon content of the Central basin was twice that of the Eastern basin. This would suggest that most of the oxygen uptake occurred in the waters of the hypolimnion. Further, an examination of the data shown in Tables 1 and 2 reveals that the areal depletion decreased with the decreasing hypolimnion mean depth, again suggesting that much of the oxygen consumption occurred within the water columns of the hypolimnia.

During the month of August 1970, a number of sediment oxygen demand studies were carried out in the Central basin (Blanton and Winklhofer 1972; Lucas and Thomas 1972). On the basis of these studies, it was concluded that approximately 80% of the Central basin hypolimnetic oxygen depletion occurred at the sediment–water interface (Burns and Ross 1972b). These findings are inconsistent with the suggestion that much of the oxygen uptake occurs in the hypolimnion water and not at the sediment–water interface. This inconsistency suggests that the topic of the site of hypolimnetic oxygen uptake requires closer scrutiny.

Since the sediment oxygen demand rate is expressed in the same units as the areal oxygen depletion rate, i.e. g O_2 m^{-2} day^{-1} and the water oxygen demand rate has the same units as the volumetric oxygen depletion rates i.e. g O_2 m^{-3} day^{-1}, it is possible to associate the areal oxygen depletion rate with the sediment oxygen demand, and the volumetric oxygen depletion rate with the water oxygen demand. This should not be done because both the areal and volumetric oxygen depletion rates are total depletion rates. These rates include the oxygen consumed at the sediment–water interface and within the water column as described in the following equations:

$$O_a = \frac{\Delta O}{A_h} = K_w \cdot d + K_s \qquad (8)$$

$$O_v = \frac{\Delta O}{V_h} = K_w + \frac{K_s}{d} \qquad (9)$$

where, O_a = areal oxygen depletion rate
O_v = volumetric oxygen depletion rate
ΔO = mean daily hypolimnetic oxygen depletion
K_w = water oxygen demand
K_s = sediment oxygen demand
d = mean hypolimnion depth
A_h = mean hypolimnion area
V_h = mean hypolimnion volume

The values for K_s and K_w can be estimated if the assumption is made that these values remained constant under oxic conditions. The mean of the values of K_s measured for the Central basin (Lucas and Thomas 1972; Blanton and Winklhofer 1972) is 0.36 g O_2 m^{-2} day^{-1}. This value, together with the data in Table 1, results in the following values being calculated for K_w in the Central basin: surveys 3 to 4, $K_w = 0.018$; surveys 4 to 5, $K_w = 0.040$ and surveys 5 to 6, $K_w = 0.006$ g O_2 m^{-3} day^{-1}. Alternatively, the data in Table 1 can be considered to give three independent equations of the form of equation (8). These equations can be solved giving three values for both K_w and K_s. The mean of these three values was found to be $K_w = 0.30$ g O_2 m^{-3} day and $K_s = -0.55$ g O_2 m^{-2} day. The difference between the values obtained by these two methods of analysis is large, especially if the measured value of K_s (0.36 g O_2 m^{-2} day^{-1}) is compared with the calculated value of K_s (which represents a production of 0.55 g O_2 m^{-2} day^{-1} from the sediments). These differences reflect the error of the initial assumption, showing that K_w and K_s are not constant with time. The data in Table 2 permits the formulation of four independent equations in the form of equation (8) which yield six values for both K_w and K_s. The mean of these values is $K_w = 0.19$ g m^{-3} day^{-1} and $K_s = -1.89$ g m^{-2} day^{-1}. This unlikely value for K_s indicates that the value for K_w was probably decreasing with the value of d, the mean hypolimnion depth, as the season progressed. In light of this information, it is not possible to comment on relative values of K_s and K_w. Both of these quantities require further investigation.

Summary and Conclusions

Increases in the Central basin hypolimnetic oxygen depletion rate over the past 40 years have provided a quantitative indicator of the degradation of Lake Erie water quality. An improvement in the water quality of the lake is anticipated as a result of a planned decrease in nutrient loadings to the lake. It is presumed that decreases in the hypolimnetic depletion rate will provide a quantitative measure of the improvement of the lake. However, because of variations induced by weather, the depletion rate will have to be measured to an accuracy of approximately 3% for about 5 consecutive years after an improvement has commenced, before a definite conclusion regarding a change in the trophic state of the lake could be reached.

The degree of reoxygenation of the hypolimnia from the mesolimnia and epilimnia of the lake can be quite variable and the magnitude of the oxygen transfer by this process must be estimated for each determination of hypolimnetic depletion rate. A mesolimnion exchange model has been described here and used in the calculation of three depletion rate values for the Central basin and four values for the Eastern basin. Further, if the hypolimnetic depletion rate is to be measured accurately, a careful measurement of the oxygen transport between the hypolimnia of the Central and Eastern basins is required.

The volumetric and areal depletion rates describe different aspects of hypolimnetic oxygen depletion and both are necessary for a full description of the phenomenon. The Central basin had an areal oxygen depletion rate (0.43 g O_2 m^{-2} day^{-1}), which was approximately half of the value of the Eastern basin rate of 0.87 g O_2 m^{-2} day^{-1}, but had a volumetric oxygen depletion rate (0.13 g O_2 m^{-3} day^{-1}) which was twice that of the Eastern basin (0.057 g O_2 m^{-3} day^{-1}).

A pattern of decreasing depletion rates associated with decreasing hypolimnion mean depths was observed and led to the supposition that much of the hypolimnetic oxygen depletion occurred in the water overlying the sediments. However, it is probable that both the oxygen depletion rate and the mean hypolimnion depth were decreasing simultaneously as the season progressed. The question of the probable site of the major oxygen uptake processes remains unresolved as a result of the simultaneous change in the two variables.

Acknowledgments

The help of Dr R. M. Baxter and Dr R. A. Vollenweider in the preparation of this manuscript is gratefully acknowledged.

BLANTON, J. O., AND A. R. WINKLHOFER 1972. Physical processes affecting the hypolimnion of the Central Basin of Lake Erie, p. 9–38. *In* N. M. Burns and C. Ross [ed.] "Project Hypo". Can. Cent. Inland Waters Pap. 6.

BURNS, N. M. 1976a. Temperature, oxygen and nutrient distribution patterns in Lake Erie, 1970. J. Fish. Res. Board Can. 33: 485–511.

1976b. Nutrient budgets for Lake Erie, 1970. J. Fish. Res. Board Can. 33: 520–536.

BURNS, N. M., F. ROSA, AND C. H. CHAN. 1976. Lake Erie water chemistry data 1970–71. Can. Cent. Inland Waters Pap. 16: 180 p.

BURNS, N. M., AND C. ROSS. 1972a. Oxygen–nutrient relationships within the Central Basin of Lake Erie, p.

85–119. *In* N. M. Burns and C. Ross [ed.] "Project Hypo". Can. Cent. Inland Waters Pap. 6: 180 p.

1972b. Project-hypo — discussion of findings, p. 120–126. *In* N. M. Burns and C. Ross [ed.] "Project Hypo". Can. Cent. Inland Waters Pap. 6: 180 p.

DOBSON, H. H., AND M. GILBERTSON. 1972. Oxygen depletion in the Central Basin of Lake Erie, 1929 to 1970, p. 3–8. *In* N. M. Burns and C. Ross [ed.] "Project Hypo". Can. Cent. Inland Waters Pap. 6: 180 p.

LUCAS, A. M., AND N. A. THOMAS 1972. Sediment oxygen demand in Lake Erie's Central Basin, 1970, p. 45–50. *In* N. M. Burns and C. Ross [ed.] "Project Hypo". Can. Cent. Inland Waters Pap. 6: 180 p.

SIMONS, T. J. 1976. Continuous dynamical calculations of water transports in Lake Erie in 1970. J. Fish. Res. Board Can. 33: (In press)

UNITED STATES DEPARTMENT OF STATE 1972. Great Lakes Water Quality Agreement with annexes and texts and terms of reference between the United States of America and Canada. U.S. Gov. Print. Off. Washington. #T1AS 7312. 69 p.

Nutrient Budgets for Lake Erie, 1970

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BURNS, N. M. 1976. Nutrient budgets for Lake Erie, 1970. J. Fish. Res. Board Can. 33: 520–536.

Budgets for the quantities of water, chloride, and phosphorus entering and leaving Lake Erie during the year of 1970 have been developed. The phosphorus budget was broken down into basin budgets which were further subdivided into epilimnion budgets for the Central and Eastern basins. Epilimnion budgets for soluble inorganic nitrogen (SIN) and soluble reactive silica (SRS) were also developed. The average coefficients of elimination by sedimentation of the phosphorus within each basin were 4.0, 0.38, and 0.18%/day for the Western, Central, and Eastern basins, respectively. The phosphorus elimination was found to be inversely related to mean basin depth but directly related to the phosphorus concentration of the water. The epilimnion budgets showed that during the summer the internal loading of SRS to the epilimnion was greater than the uptake of SRS by phytoplankton growth. The epilimnion budget of SIN demonstrated a massive uptake of the material during the summer, which was only partially replenished. The loss rate of phosphorus from the Central and Eastern basin epilimnia decreased as the summer progressed, to the extent that there was a net gain of phosphorus in the Eastern basin epilimnion toward the end of the summer. It is believed that this resulted from significant upward transport of phosphorus from the hypolimnion to the epilimnion by flagellate species of phytoplankton.

BURNS, N. M. 1976. Nutrient budgets for Lake Erie, 1970. J. Fish. Res. Board Can. 33: 520–536.

Nous avons calculé les budgets d'eau, de chlorures et de phosphore pénétrant dans le lac Erié et en sortant durant l'année 1970. Le budget de phosphore a été divisé en budgets par bassin et subdivisé davantage en budgets épilimnétiques pour les bassins central et oriental. Nous avons également établi les budgets épilimnétiques pour l'azote inorganique soluble (NIS) et la silice réactive soluble (SRS). Les coefficients moyens d'élimination du phosphore par sédimentation dans chaque bassin sont de 4.0, 0.38 et 0.18% par jour pour les bassins occidental, central et oriental respectivement. L'élimination du phosphore est inversement reliée à la profondeur moyenne du bassin, mais directement reliée à la teneur en phosphore de l'eau. Les budgets épilimnétiques démontrent qu'en été, l'apport interne de SRS à l'épilimnion est plus grand que l'assimilation de SRS par croissance du phytoplancton. Le budget épilimnétique de la NIS démontre qu'il y a assimilation massive du matériel en été et qu'il n'est que partiellement renouvelé. Le taux de perte du phosphore de l'épilimnion des bassins central et oriental diminue à mesure que l'été avance, au point qu'il y a gain net de phosphore dans l'épilimnion du bassin oriental vers la fin de l'été. On croit que ceci résulte d'un transport important de phosphore vers le haut, de l'hypolimnion vers l'épilimnion, par les flagellés du phytoplancton.

Accepted November 18, 1975 Accepté le 18 novembre 1975

THE most effective manner in which to initiate the quantitative study of the chemical limnology of a lake is by budget studies. The calculations reveal the magnitude of the net transfers of materials between the various compartments of a lake. Once the pattern of net material transports between these compartments has been established, it becomes possible to investigate and find the predominant mechanisms by which these material transports occurred (Vollenweider 1976). The finer the scale, both in time and space, on which the budget studies are carried out, the more precise does the subsequent investigation of transfer mechanisms become.

In this report, the results of the annual total lake budgets for water, the chloride ion, and phosphorus are first presented. The next consideration is the calculation of the quantities of chloride and phosphorus which were present in the lake during the 10 lakewide surveys carried out in 1970 by the M.V. *Martin Karlsen*. By noting the concentrations of the elements of interest at the basin boundaries and also the quantities of these elements in each basin, it was possible to construct basin budgets for each of the intervals between surveys. The basin budgets were subsequently utilized in the construction of

Printed in Canada (J4084)
Imprimé au Canada (J4084)

epilimnion budgets for phosphorus in the Central and Eastern basins for four survey intervals during the summer.

Although the data on the nitrogen and silica inputs to the lake are less precise than those for the phosphorus inputs, it has been possible to estimate epilimnion budgets for soluble inorganic nitrogen and soluble reactive silica.

Budget Equations

The equations formulated here are simple and provide the most concise means of describing the different types of budget calculations carried out in this study.

The equation used in the water budget calculations is,

$$\Delta V_e = \sum_i V_R - \sum_i V_O \pm \Delta S \pm G \qquad (1)$$

where ΔV_e = monthly change in volume due to net evaporation or precipitation
V_R = monthly volume of river inputs
V_O = monthly volume of river outputs
ΔS = monthly change in volume of water storage
G = monthly groundwater flow
≈ 0.0 (by assumption)

The chloride ion is considered to be conservative within the Lake Erie system. During 1970 there was no discernible change in the mean chloride concentration of the lake; thus the following equations were used in the chloride budget studies:

$$Q_{ci} = Q_{co} + \Delta S_c \qquad (2)$$

where Q_{ci} = annual chloride input
Q_{co} = annual chloride output
ΔS_c = annual change in chloride storage

Also,

$$Q_{cn} = Q_{c(n-1)} + \sum_i (V_r C_{rc})_{n,(n-1)} - \sum_i (V_o C_{oc})_{n,(n-1)} + (D_c)_{n,(n-1)} + (V_p C_{pc})_{n,(n-1)} \qquad (3)$$

where Q_{cn} = quantity of chloride within the lake during survey n
$Q_{c(n-1)}$ = quantity of chloride in lake during survey $n-1$
$(V_r C_{rc})_{n,(n-1)}$ = river input of chloride between surveys n, $n-1$
$(V_o C_{oc})_{n,(n-1)}$ = river output of chloride between surveys n, $n-1$
$(D_c)_{n,(n-1)}$ = direct discharge of chloride to the lake between surveys
$(V_p C_{pc})_{n,(n-1)}$ = precipitation input of chloride between surveys

Phosphorus is a nonconservative substance in the lake system and the equation used for the phosphorus lake budget is,

$$E_p = Q_{pn} - \sum_i (V_r C_{rp})_{n,(n-1)} + \sum_i (V_o C_{op})_{n,(n-1)} - (D_p)_{n,(n-1)} - (V_p C_{pp})_{n,(n-1)} - Q_{p(n-1)} \qquad (4)$$

where E_p = phosphorus eliminated from or regenerated to the waters of the lake between surveys n and $n-1$ (budget imbalance)
Q_{pn} = quantity of phosphorus in the lake at the time of survey n
$Q_{p(n-1)}$ = quantity of phosphorus in the lake at the time of survey $n-1$
$(V_r C_{rp})$ = river inputs of phosphorus between surveys
$(V_o C_{op})$ = river output of phosphorus from the lake between surveys
$(V_p C_{pp})_{n,(n-1)}$ = inputs of phosphorus by precipitation between surveys
$(D_p)_{n,(n-1)}$ = direct discharge of phosphorus to the lake between surveys

The three phosphorus basin budgets were calculated like the lake budget except that because of the west to east flow pattern, the outflow from the Western basin became an input for the Central basin and the Central basin output became an input for the Eastern basin.

Before any nutrient budget can be constructed, the decision must be made whether to use mass budgets (Vollenweider 1975; Imboden 1974) or net budgets (Johnson and Owen 1971). If a river brings a large quantity of a nutrient into a lake at a lower concentration than the nutrient is present in the lake, the river will act as a diluting factor and its net input into the lake will be negative. In the case of a mass budget, the quantity loaded into the lake is considered independent of the concentration and the loading is always positive. Mass budgets are easier to use when the behavior of the nutrients within the lake is the primary consideration. Net budgets are preferable when the nature of the different input sources is being examined, as follows later in this presentation.

The simple equation for estimating the loss or gain from the lake water of a nonconservative substance x, between the surveys is:

$$\Delta x = V_2 C_2 - V_1 C_1 - V_a C_a - V_b C_b + V_o C_o \qquad (5)$$

where V_1 = volume of water mass during Survey 1
V_2 = volume of water mass during Survey 2
V_a = volume of water added from source a between surveys
V_b = volume of water added from source b between surveys
V_o = volume of water discharged between surveys

and C refers to the concentrations of the nutrient in the different water masses. The volume of the water is considered to be conservative,

$$V_2 - V_1 = V_a + V_b - V_o \quad (6)$$

$$\text{and } \Delta x = V_2 (C_2 - C_1) - V_a (C_a - C_1) - V_b (C_b - C_1) + V_o (C_o - C_1) \quad (7)$$

Each of the terms on the right-hand side of the equation (7) gives the net budget contribution of the individual sources when the reference concentration is taken as the concentration of the water mass during the first survey of any pair of surveys.

The equation for the epilimnion net budgets of phosphorus in the Central basin then becomes,

$$\Delta P = \underbrace{V_2(C_{p2} - C_{p1})}_{\text{observed change in quantity}} - \underbrace{V_r(C_{rp} - C_{p1})}_{\text{input from rivers and direct discharges}} - \underbrace{V_{ow}(C_{pw} - C_{p1})}_{\text{input from Western basin}} + \underbrace{V_{oe}(C_{pe} - C_{p1})}_{\text{loss to Eastern basin}} - \underbrace{V_p(C_{pp} - C_{p1})}_{\text{input from precipitation}} - \underbrace{V_h(C_{hp} - C_{p1})}_{\text{input by hypolimnion incorporation}} - \underbrace{P_{ex}}_{\text{input from hypolimnion by exchange}} \quad (8)$$

where ΔP = input of phosphorus to the epilimnion by mechanisms other than those listed above

C_{p1} = concentration of phosphorus during survey 1

C_{p2} = concentration of phosphorus during survey 2

V_r = volume of river and direct discharge inputs between surveys

C_{rp} = mean concentration of phosphorus in rivers and municipal discharges between surveys

V_{ow} = volume of water flowing from the Western into the Central basin between surveys

C_{pw} = concentration of phosphorus at Western-Central basin epilimnion boundary between surveys

V_{oe} = volume of water flowing from the Central into the Eastern basin between surveys

C_{pe} = concentration of phosphorus at the Central to Eastern basin epilimnion boundary between surveys

V_p = volume of precipitation between surveys

C_{pp} = concentration of phosphorus in precipitation

V_h = volume of hypolimnion water incorporated into epilimnion

C_{hp} = mean concentration of phosphorus in water incorporated from hypolimnion to epilimnion

P_{ex} = quantity of phosphorus added to the epilimnion from the hypolimnion by the exchange mechanism

The Eastern basin epilimnion budgets were calculated by using a corresponding equation. Equation (8) can be rewritten as follows so that the effect of each source on the mean phosphorus concentration of the basin can be calculated;

$$\frac{\Delta P}{V_2} = (C_{p2} - C_{p1}) - \frac{V_r}{V_2}(C_{rp} - C_{p1}) - \frac{V_{ow}}{V_2}(C_{pw} - C_{p1}) + \frac{V_{oe}}{V_2}(C_{pe} - C_{p1}) - \frac{V_p}{V_2}(C_{pp} - C_{p1}) - \frac{V_h}{V_2}(C_{hp} - C_{p1}) - \frac{P_{ex}}{V_2} \quad (9)$$

The magnitude of these concentration changes can be compared with the mean concentration actually present in the basin so that the relative effect of each source's loading on the phosphorus in the basin can be observed.

Methods and Results

Water Budget

As a first step in obtaining a realistic materials loading to the lake, it was considered necessary to obtain a water budget that was essentially correct. Equation (1) was used and the results are shown in Table 1. Flows for the major streams and rivers flowing into Lake Erie were obtained from published records (U.S. Geological Survey 1970; Water Survey of Canada 1970) for the development of the water budget. The storage was calculated from documented changes in water level (Marine Sciences Directorate 1973). The imbalance in the monthly budgets was considered to be due to the net difference between precipitation and evaporation, and the appropriate change in lake height was calculated. The precipitation was found to be greater than the evaporation in the spring with the reverse being

Table 1. Summary of water budget for Lake Erie in 1970 ($km^3\ yr^{-1}$).

Detroit River Input	183.267
Western basin rivers	5.683
Central basin rivers	4.228
Eastern basin rivers	3.297
Total Input	196.475
Net Storage	+3.06
Estimated Output	193.39
Measured Output	187.88
Difference	5.51
Net evaporation due to difference ($cm\ yr^{-1}$)	21.9

TABLE 2. Hydrologic and morphometric data for Lake Erie.

	Total lake	Western basin	Central basin	Eastern basin
Surface area (km^2)	25,212	3,680	15,390	6,150
Volume (km^3)	468	28	274	166
Mean depth (m)	18.6	7.6	17.8	27.0
Maximum depth (m)	64	14	26	64
1970 Flow-through ($km^3 yr^{-1}$)	197	189	193	197
Mean water residence time (yr)	2.4	0.2	1.4	0.8

the case in the fall; the annual evaporation was estimated as being greater than annual precipitation, which is in agreement with the average pattern (Phillips and McCulloch 1972).

Data on the basic morphometric and hydrologic properties of the lake are shown in Table 2.

LAKEWIDE CHLORIDE AND PHOSPHORUS BUDGETS

The loadings of chloride and phosphorus by river were calculated by multiplying the integrated continuous flow data of a river for a month by the mean monthly concentration of the parameter as measured by weekly or biweekly river sampling programs (U.S. Geological Survey 1970, 1967; Ontario Ministry of the Environment, 135 St. Clair Avenue West, Toronto, Ont. personal communication). Precipitation estimates of chloride and phosphorus were obtained from precipitation data collected by samplers on Pelee Island, Port Stanley, and Long Point during 1970 (M. T. Shiomi personal communication). Nontributary loadings (direct discharges to the lake) were estimated from the values listed in the report to the International Joint Commission (International Lake Erie Water Pollution Board 1969). Precipitation loadings of nitrogen and silica were estimated from some summer measurements of concentrations of NO_3^- plus NH_3 and SiO_2 in rainfall collected on board a ship in Lake Erie (D. H. Matheson unpublished data) together with average summer rainfall quantities (Philips and McCulloch 1972). The estimated chloride and phosphorus loadings to the lake are shown in Tables 3 and 4. Loadings, detailed by river, are published elsewhere (Burns et al. 1976).

Ongley (1976) has raised real doubts as to the validity of computing river loadings by multiplying the values obtained for monthly mean flow and monthly mean concentration. However, it appears that the methods used for computing river loadings in this study were adequate since the imbalance in the annual chloride budget (Table 3) represents only 3.8% of the annual budget. Since phosphorus is nonconservative in the waters of a lake, it is not possible to calculate the accuracy of a phosphorus budget directly. It is assumed here that the whole lake phosphorus budget has an accuracy similar to that of the annual chloride budget.

TABLE 3. Summary of chloride budget for Lake Erie in 1970 (metric tons × 10^3 of Cl^-).

Detroit River Input	3530.7
Western basin rivers, discharges and precipitation	184.5
Central basin rivers, discharges and precipitation	865.3
Eastern basin rivers, discharges and precipitation	72.2
Total Input	4652.7
Storage (mean concn is 25.0 mg Cl $liter^{-1}$)	76.5
Budget output (estimate)	4576.2
Measured output	4757.1
Imbalance	180.9
% Imbalance	= 3.8%

TABLE 4. Summary of phosphorus budget for Lake Erie in 1970 (metric tons of P).

Detroit River input	18,075
Western basin rivers, discharges and precipitation	5,723
Central basin rivers, discharges and precipitation	5,769
Eastern basin rivers, discharges and precipitation	2,494
Total input	32,061
Total output	3,843
Phosphorus net loading	28,218
Retention within the lake	88.0%

Ongley (1976) shows detailed estimates of the mean annual input of chloride and total phosphorus to Lake Erie from the northshore rivers (excluding the Detroit River) averaged over the years 1964–72. Ongley's estimate for annual mean loading of chloride to Lake Erie from the north shore is 86,300 metric tons/yr which com-

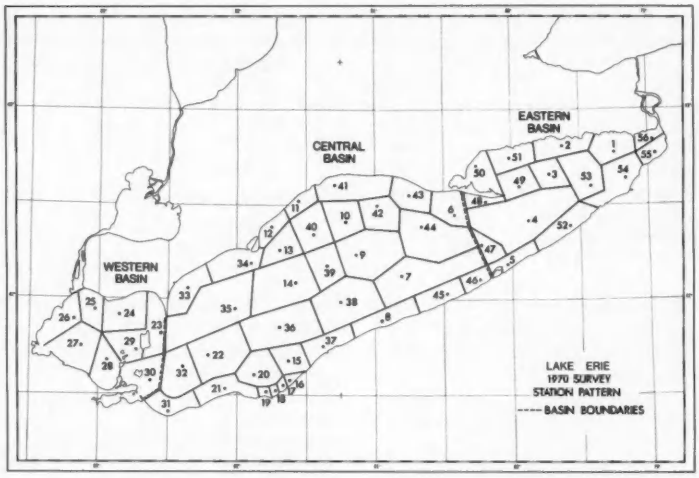


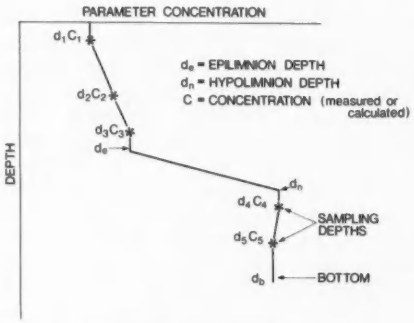
FIG. 1. Lake Erie water sampling stations showing area-weighting factors and basin boundaries.

pares with the loadings of 64,000 metric tons from the northshore tributaries (excluding the Detroit River) included in Table 3. The northshore tributary loading of phosphorus included in Table 4 amounts to 1644 metric tons which compares with Ongley's estimate of the mean annual loading of 1564 metric tons. These values indicate basic agreement between the data sources.

ESTIMATION OF THE QUANTITIES OF MATERIALS IN LAKE ERIE

A description of the sampling methods used and a list of the variables measured during the 1970 M.V. *Martin Karlsen* ship surveys is given by Burns (1976a) and the sampling station pattern is shown in Fig. 1. Burns (1976a) also gives reasons for the lake being considered as a single, vertically well-mixed layer, the epilimnion, during the unstratified period. During the stratified period, the lake appears to consist of two vertically well-mixed layers, the epilimnion and hypolimnion separated by a pycnocline layer, the mesolimnion, which is also a chemocline layer. The computational scheme used to calculate the depth-weighted mean station concentrations for each layer at each station is shown in Fig. 2.

Since higher concentrations were often observed at the shallower shoreline stations than at the deeper offshore stations, the simple basin mean concentration for a species was significantly higher than the volume-weighted basin mean. Thus a volume-weighting scheme was used for



$$C_{EPILIMNION} = \frac{1}{d_e}\left(d_1C_1 + (d_2-d_1)\times\tfrac{1}{2}(C_1+C_2) + (d_3-d_2)\times\tfrac{1}{2}(C_2+C_3) + (d_e-d_3)C_3\right)$$

$$C_{MESOLIMNION} = \tfrac{1}{2}(C_3+C_4)$$

$$C_{HYPOLIMNION} = \frac{1}{(d_b-d_n)}\left((d_4-d_n)C_4 + (d_5-d_4)\times\tfrac{1}{2}(C_4+C_5) + (d_b-d_5)C_5\right)$$

FIG. 2. A diagramatic illustration of the depth weighting scheme used.

the calculation of the quantity of a species in each basin. An area was assigned to each sampling station, as shown in Fig. 1, which took into account the area for which the station could be considered to be representative. The volume below that area was calculated for each thermal layer to obtain the volume weighting factor for each layer at each station. Epilimnion and hypolimnion basin quantities were obtained by carrying out the appropriate summations.

TABLE 5. Comparison of observed and estimated quantities of chloride (units = metric tons of Cl^- × 10^6)

Survey dates 1970	Survey no.	Estimated quantity		Observed quantity	Difference[a] (Observ.–Est.)
Apr 7–11	1	Quantity	11.452	11.452	
		In	.317		
		Out	.299		
May 6–10	2	Est.Q.(2)	11.470		
		In	.339		
		Out	−.433		
June 2–6	3	Est.Q.(3)	11.376	11.883	+.507
		In	.430		
		Out	−.402		
July 3–7	4	Est.Q.(4)	11.861	11.493	−.325
		In	.363		
		Out	−.416		
July 28–Aug. 2	5	Est.Q.(5)	11.483	11.473	−.010
		In	.391		
		Out	−.405		
Aug. 25–30	6	Est.Q.(6)	11.459	11.645	+.186
		In	.369		
		Out	−.394		
Sept. 23–27	7	Est.Q.(7)	11.620	11.456	−.164
		In	.381		
		Out	−.400		
Oct. 21–26	8	Est.Q.(8)	11.437	11.332	−.105
		In	.394		
		Out	.351		
Nov. 25–30	9	Est.Q.(9)	11.375	11.288	−.087
		In	.248		
		Out	.214		
Dec. 14–18	10	Est.Q.(10)	11.322	11.447	+.115

[a]SD of difference = ± 1.58 = 1.3% lake content of chlorine. Therefore, accuracy of ship survey = ±2.6% of quantity calculated.

Besides validating the river input–output budgets, the chloride budget also permitted estimates to be made of the precision of the values calculated for the quantities of materials present in the lake at any one time. Nine surveys of the quantity of chloride in Lake Erie were included in the 1970 investigations. These surveys gave values which were inserted into equation (3) for eight different survey intervals; the results and the survey dates are shown in Table 5. The standard deviation of the eight differences between the values on the left- and right-hand sides of equation (3) was approximately 1.3% of the quantity of the chloride in the lake (Table 5). This value is considered to be an indication of the precision of the quantity estimates calculated for any material sampled on a lakewide basis during the 1970 M.V. *Martin Karlsen* surveys. Thus the values calculated for quantities present in the lake are of an accuracy comparable with that of the lake budgets which were calculated for phosphorus and the chloride ion.

BASIN BUDGETS

The calculation of budgets for the Western, Central, and Eastern basins required that the transport of phosphorus, soluble inorganic nitrogen (SIN), and soluble reactive silica (SRS) between the basins be estimated. This was done using the water transport quantities calculated by

TABLE 6. Summary of transport of phosphorus between basins in Lake Erie (metric tons of phosphorus). (Positive values represent west to east transport).

	Transect 1 (Western basin to Central basin)			Transect 2 (Central basin to Eastern basin)		
Survey interval	Diffusive transport	Advective transport	Total transport	Diffusive transport	Advective transport	Total transport
1–10	91	3512	3603	−15	1896	1881
10 to 1–71 (est.)	22	2589	2611	0	1351	1351
1–70 to 1–71	113	6101	6214	−15	3247	3232

Simons (1976) for both the epilimnion and hypolimnion across two transects in the lake. The transport from the Western basin to the Central basin was calculated using the transect which runs south from Point Pelee at right angles to the main axis of the lake. The transect at which the transports between the Central and Eastern basin were calculated runs northward from Erie, Pennsylvania, at a right angle to the main axis of the lake.

The quantity of oxygen, soluble inorganic nitrogen, and soluble reactive silica transported between two surveys from the Western basin to the Central basin was estimated by multiplying the flow by the mean value of the epilimnion concentrations observed at stations 23 and 32 during both surveys. The epilimnion transport from the Central to the Eastern basin was similarly calculated using the mean of the epilimnion concentrations at stations 6, 47, and 46. The hypolimnion transport between these two basins was estimated on the basis of the concentrations observed at station 47. Diffusive transports of the above materials between basins have not been calculated because they are significantly smaller than the advective transports and the estimates of the external inputs of nitrogen and silica to these basins are not sufficiently accurate to require the calculation of diffusive transports.

The transport of phosphorus between the basins was calculated more precisely as the external input data are more accurate. Each of the transects described above traversed eight cells and the flow was computed through each of these cells for each survey interval (see Simons 1976). The concentration in each of the eight cells and the concentration gradient in the east–west direction through each cell was obtained from total phosphorus distribution maps which had values extrapolated over 596 grid points as described by Lam and Simons (1976). The transport of phosphorus through each cell by advection was estimated by multiplying the flow by the concentration; the diffusive transport was estimated using a diffusion coefficient of 2×10^5 cm^2 s^{-1} (Lam and Simons 1976). The results of these computations are shown in Table 6.

A survey of Lake Erie was carried out during April 1971 and showed the quantities of phosphorus in the lake to be very similar to those observed during April 1970. To obtain annual basin budgets it was assumed that the phosphorus loadings to the lake and flow patterns in the lake were the same from January to April 1971 as during the corresponding period in 1970.

The basin budgets of total phosphorus (Table 7) are important because they reveal the periods of net phosphorus regeneration and sedimentation quantitatively, making it possible to calculate net flux rates of phosphorus to or from the sediments. The quantities of phosphorus eliminated from the water or regenerated to the water were computed using equation (4) for each of the basins and are shown in Table 7, with the rates calculated from these values shown in Fig. 3.

EPILIMNION BUDGETS

The calculation of the values for the first two terms on the right-hand side of equation (9) is obvious. The concentration values used in calculating the third term are the mean of the values measured at stations 23 and 32, and the concentration values used in calculating the fourth term are the mean of the values observed at stations 6, 46, and 47. The last two terms of equation (9) are more difficult to calculate but are necessary for the construction of an epilimnion budget. The mesolimnion exchange model (Burns 1976b) was used for the calculation of both of these terms. Burns (1976b) used this model for the estimation of oxygen transferred from the mesolimnion to the hypolimnion and indicated how the model can be used to estimate the transport of materials from the hypolimnion to the mesolimnion and/or epilimnion. The epilimnion has been considered in this case to consist of both the mesolimnion and epilimnion for the budget

TABLE 7. Imbalance in Lale Erie phosphorus basin budgets (metric tons of phosphorus). Negative value indicates loss of phosphorus from water.

Survey interval	Western basin	Central basin	Eastern basin	Lake total
1–2	− 2525	+ 828	− 951	− 2648
2–3	− 1235	− 790	+ 273	− 1752
3–4	− 1768	−2024	− 593	− 4385
4–5	− 1585	−1781	− 228	− 3594
5–6	− 561	+ 22	− 795	− 1334
6–7	− 2050	+ 983	+ 267	− 800
7–8	− 1169	− 684	+ 37	− 1816
8–9	− 1013	+2816	+1390	+ 3193
9–10	− 1126	+ 783	− 29	− 372
1–10 Total	−13032	+ 153	− 629	−13508
10 to 1–71	− 5152	−7906	−1339	−14397
Annual total imbalance	−18184	−7753	−1968	−27905
Annual total imput	23583	11947	5707	
% Removal from water	77%	65%	35%	

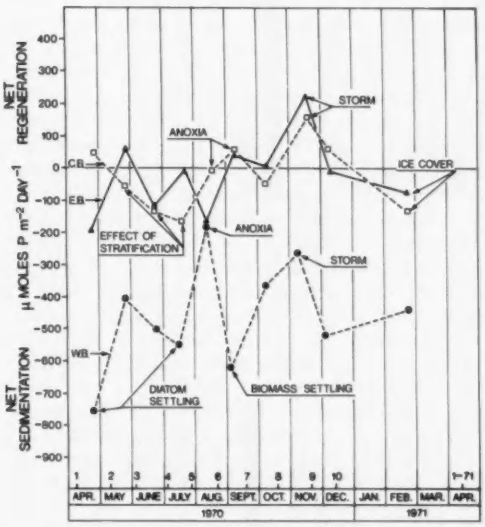


FIG. 3. Net sedimentation rates of phosphorus observed in the Western, Central, and Eastern basin of Lake Erie.

calculations. This has been done since the mesolimnion can often be within the photic zone and because its thickness can vary greatly, with the nutrients transferred up from the hypolimnion being held only temporarily within the mesolimnion.

The results of the epilimnion net budget calculations for SRS, $NH_3 + NO_3^- + NO_2^-$, and total phosphorus are shown in Fig. 4(a–f). Detailed results, giving values for each source during each survey interval from survey 3 to 7 are published elsewhere (Burns et al. 1976). Epilimnion distribution maps for total phosphorus are shown in Fig. 5.

Discussion

There is a large input of phosphorus to the Central and Eastern basins due to massive erosion of the unconsolidated bluffs along the north shore of the Central basin (Kemp et al. 1976). However, the input of phosphorus from the bluffs is almost entirely in the form of apatite which apparently does not dissolve in Lake Erie (Williams et al. 1976). An examination of the total phosphorus distribution maps reveals that the waters in the northeastern part of the Central basin, which receive the largest erosional loadings of phosphorus, often contain the lowest levels of total phosphorus (see Fig. 5 and Burns 1976a). Because of the inertness of most of the phosphorus of erosional origin, it is considered reasonable to restrict the discussion of processes in the lake to the phosphorus of nonerosional origin. Further, since the changes between the various chemical forms of phosphorus in lakes can be in the order of minutes (Lean 1973), it is futile to compute budgets of the individual phosphorus forms; thus the phosphorus budgets considered here are in terms of total phosphorus only.

BASIN BUDGETS

The calculation of basin budgets (Table 7) has enabled the estimation of the net sedimentation

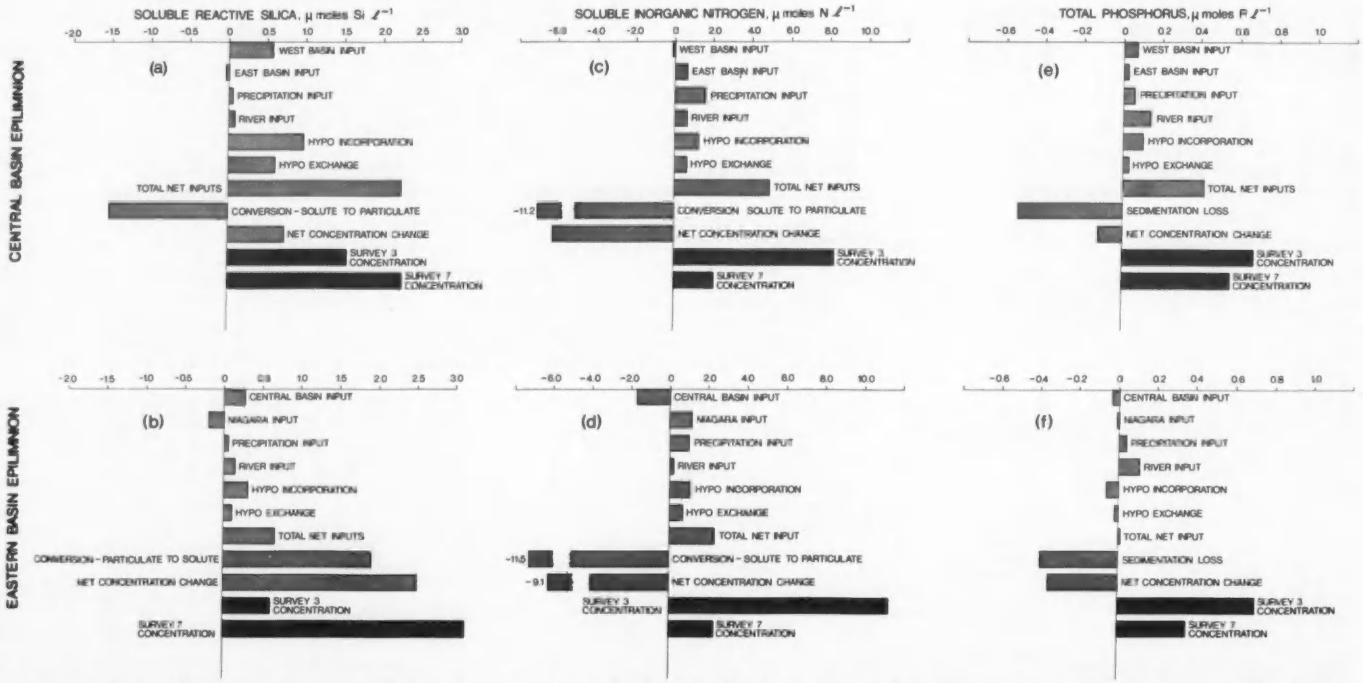


FIG. 4. Net epilimnion budgets showing the changes in concentration due to the various budget components. a, Soluble Reactive Silica in the Central basin; b, Soluble Reactive Silica in the Eastern basin; c, Soluble Inorganic Nitrogen in the Central basin; d, Soluble Inorganic Nitrogen in the Eastern basin; e, Total Phosphorus in the Central basin; and f, Total Phosphorus in the Eastern basin.

TABLE 8. Mean rates of phosphorus elimination processes in the basins of Lake Erie.

	Western basin	Central basin	Eastern basin
Net sedimentation rate (*mmol P m^{-2} day^{-1}*)	.437	.045	.028
Mean total phosphorus concn (*mmol P m^{-3}*)	1.43	.66	.57
Mean particulate phosphorus concn (*mmol P m^{-3}*)	.94	.40	.31
Mean basin depth (*m*)	7.6	17.8	27.0
Mean elimination coefficient[a] (*% day^{-1}*)	4.02	.38	.18
Mean net settling velocity[b] of particulate phosphorus (*m day^{-1}*)	.46	.11	.09

[a]Mean elimination coefficient = (sedimentation flux)/(depth) × (mean concn.).
[b]Mean net settling velocity = (sedimentation flux)/(mean particulate concn.).

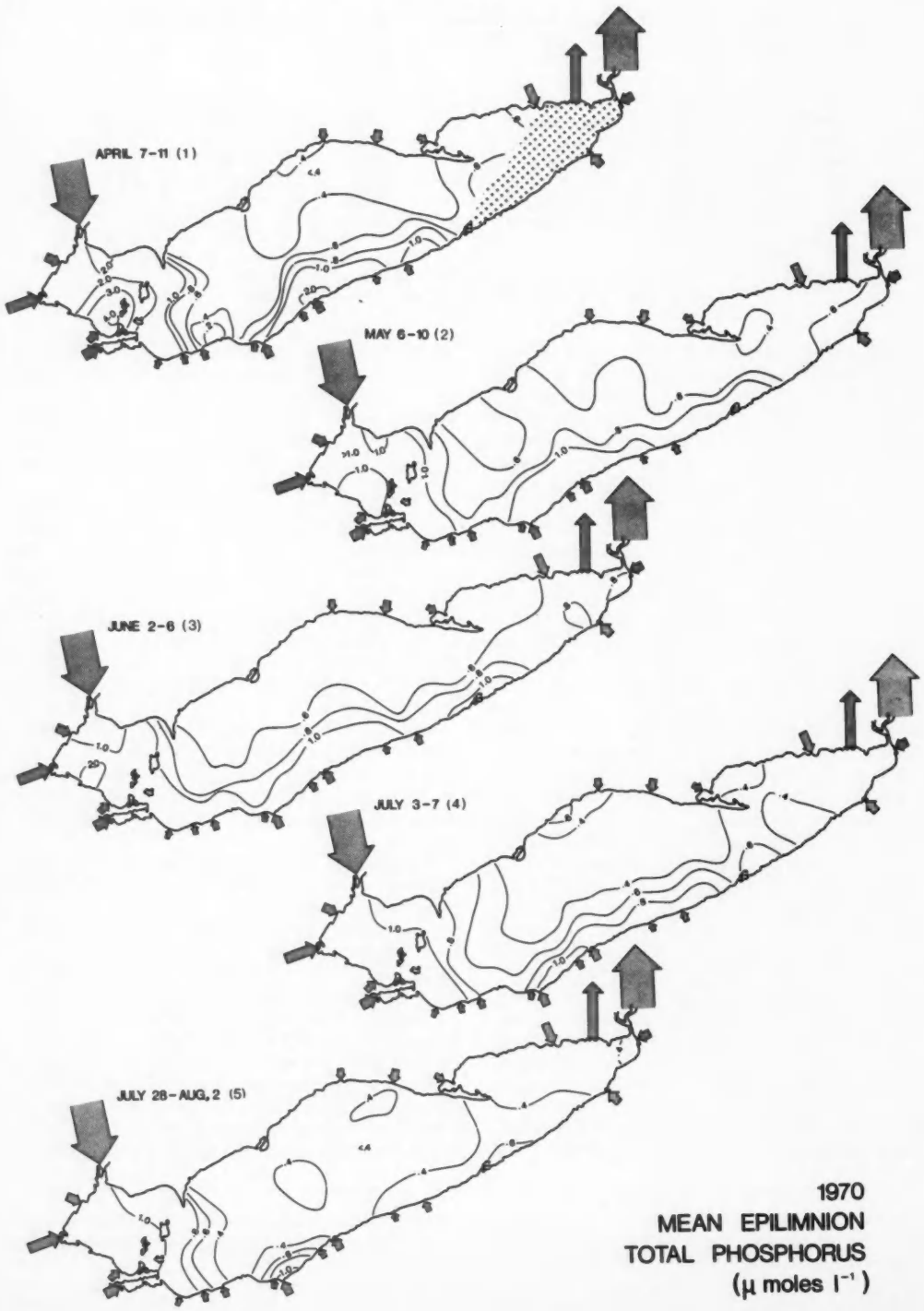
rate of phosphorus from the lake water for each cruise interval. Regeneration from the sediments and settling of phosphorus to the sediments probably occur simultaneously over the majority of each basin; however, at any one time, regeneration or sedimentation can be strongly predominant and a reason ascribed to the predominance.

The sedimentation–regeneration processes are more obvious in the Western basin than in the other two basins (Fig. 3) with net sedimentation predominating at all times and having a mean value of 437 μmol P m^{-2} day^{-1}. Dividing the mean net sedimentation rate by the mean particulate phosphorus concentration gives an approximate mean settling velocity of 0.46 m day^{-1} for the 12-mo period from April 1970 to April 1971 in the Western basin (see Table 8). The high sedimentation rate in the Western basin between surveys 1 and 2 (Fig. 3) can be ascribed to the crash of a diatom population (Munawar and Munawar 1976) with subsequent settling out. The net settling velocity of particulate phosphorus during this period is estimated at 0.75 m day^{-1}. A similar phenomenon appears to have occurred between surveys 4 and 5 with the net settling velocity of the phosphorus being about 0.73 m day^{-1}.

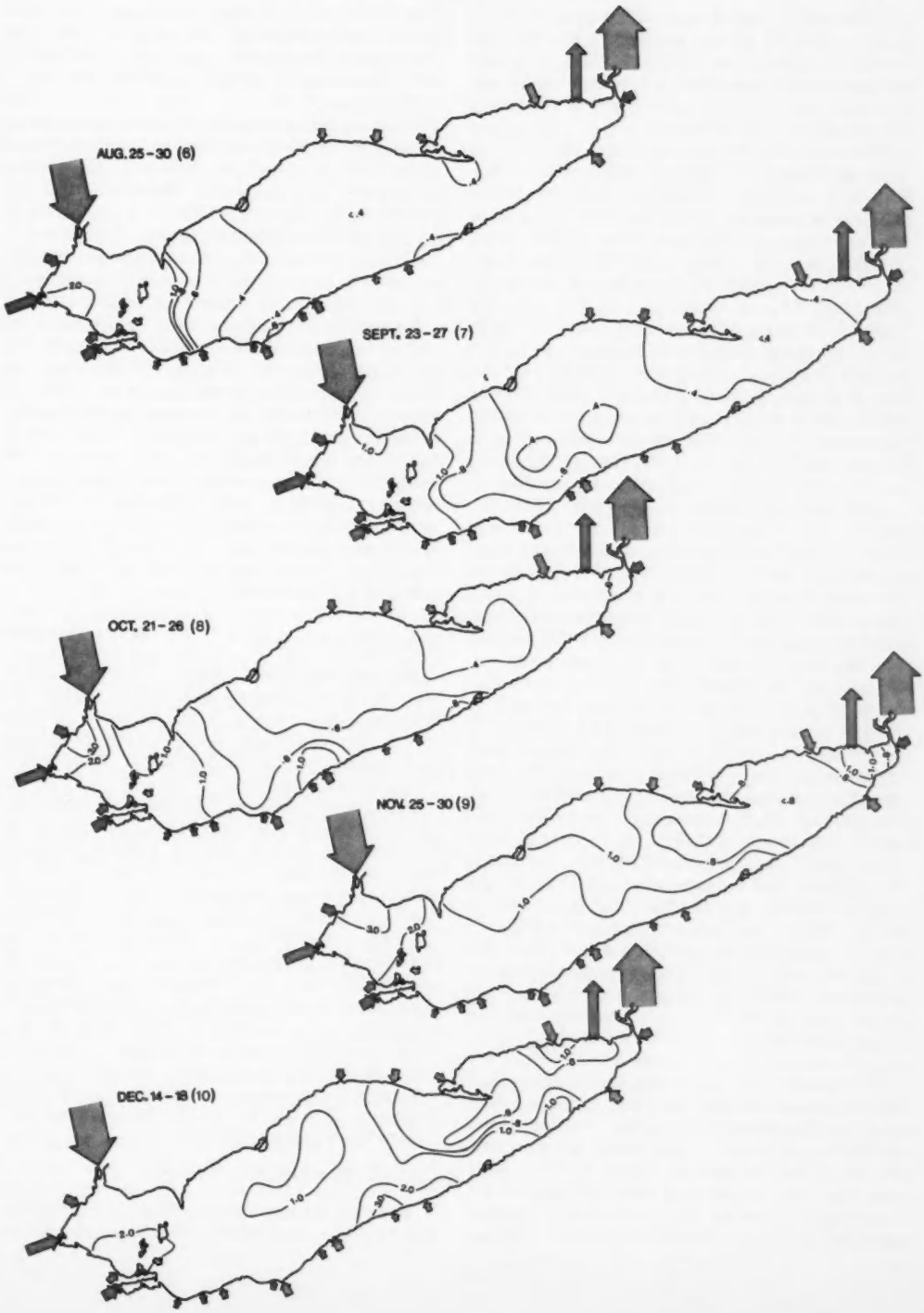
It is difficult to account for the dramatic decrease in net sedimentation in the interval between surveys 5 and 6. However, this period was characterized by an absence of strong winds with the lakewide mean wind velocity never being greater than 6 m s^{-1} (T. J. Simons, Canada Centre for Inland Waters, personal communication). Anoxic conditions have been observed in the bottom waters of the Western basin (Britt 1955) and from a recent study (Kramer et al. 1970) it can be deduced that in 1969 less than 3 days of calm weather (winds less than 5 m s^{-1}) could produce anoxic conditions in these bottom waters. Thus it is suggested that the low phosphorus net sedimentation rate during this calm period was due to high phosphorus regeneration resulting from anoxic conditions at the sediment–water interface. The net sedimentation rate was reduced to 176 from 588 μmol P m^{-2} day^{-1} (mean value for the periods between surveys 4 to 5 and 6 to 7) suggesting a release rate of 412 μmol P m^{-2} day^{-1} due to anoxic conditions. This compares with the anoxic phosphorus release rate of 240 μmol P m^{-2} day^{-1} measured in the Central basin (Burns and Ross 1972). During the period between surveys 8 and 9 there was a major storm (Lam and Jaquet 1976) which caused the Western basin mean turbidity to increase from 3.9 Jtu (survey 8) to 19.5 Jtu (survey 9). This storm caused an increase in the regeneration rate of phosphorus and thus a subsequent decrease in the net sedimentation rate.

The Central basin phosphorus processes are similar to those of the West basin except that the phosphorus sedimentation is occasionally less than the regeneration and can result in the phenomenon of a net regeneration of phosphorus (Fig. 3, Table 7). In the interval between surveys 2 and 5 there was a period of net sedimentation with the net settling velocities of particulate phosphorus increasing from 0.15 to 0.68 m day^{-1}. This coincided with the onset of thermal stratification, which probably decreased the hypolimnion vertical turbulence, permitting the net settling velocity to increase. Between surveys 5 and 7 anoxic conditions were present in the Central basin (Burns

FIG. 5. Lake Erie mean epilimnion total phosphorus concentrations, April–December 1970. The arrows depict the positions of water flow in and out of the lake with the width of the arrows being proportional to the average water flows. Survey numbers are denoted in parentheses. (p. 530–31).



1970
MEAN EPILIMNION
TOTAL PHOSPHORUS
(μ moles l^{-1})


AUG. 25 - 30 (6)
SEPT. 23 - 27 (7)
OCT. 21 - 26 (8)
NOV. 25 - 30 (9)
DEC. 14 - 18 (10)

1976a) which resulted in a net regeneration of phosphorus. The stormy weather during November and December caused a further period of net regeneration between surveys 8 and 10 which was followed by a period of net sedimentation during winter when ice cover existed.

The reasons for the variability in the Eastern basin sedimentation–regeneration phenomena are not easily understood. However, the process of phosphorus regeneration by the November storm (Lam and Jaquet 1976) dominated the effects of all other processes occurring at that time in the basin as can be seen by the net regeneration of phosphorus (Fig. 3, Table 7).

The mean phosphorus removal rates for each basin are shown in Table 8. It can be seen that the sedimentation rate differs by an order of magnitude between the Western and Central basins; the average coefficient of elimination by sedimentation of the phosphorus content of each basin is 4.02, 0.38, and 0.18%/day for the Western, Central, and Eastern basins, respectively.

The mean net sedimentation rates of phosphorus are inversely related to depth and are directly related to the concentration of the total phosphorus in water (Table 8). The decrease in the mean net settling velocity of particulate phosphorus with depth is unexpected in that in shallow waters there is greater turbulence near the bottom than in deeper waters, causing resuspension of settled materials which would result in a lower net settling velocity. However, the net settling velocity of phosphorus appears to be increased in shallow water. This apparent contradiction could be understood by considering that the phosphorus within the organic matter, which is decomposing after resuspension from the sediment surface, will probably be in contact with resuspended clay particles and the associated iron oxides (Burns and Nriagu 1976) which strongly adsorb phosphorus (Williams et al. 1976). These iron oxides on settling would remove a large part of the phosphorus from the lake water, thus increasing the net settling rate of phophorus. The effective scavenging of phosphorus from the water by suspended clay particles can be expected to increase with increasing phosphorus concentration as shown (Table 8).

It is worth noting that while the Eastern basin phosphorus net sedimentation rate is much lower than that of the Central basin, the mean net settling velocities of the particulate materials in the two basins are similar (Table 8). This indicates that the low net sedimentation rate in the east is due to a smaller fraction of the phosphorus being in the particulate form (Burns 1976a). The trend of decreasing elimination of phosphorus with increasing depth suggests that extra care must be taken in determining safe loadings of phosphorus to deeper lakes (Vollenweider 1975).

It is possible to speculate on the water quality conditions to be found in the Western basin, should the International Water Quality Agreement (U.S. Department of State 1972) be put into effect, i.e. reduction of the present levels of total input of phosphorus (approx. 32,000 metric tons) to the whole lake from rivers and outfalls to 14,600 metric tons by 1976. The Western basin would show a definite improvement within a few years reflecting the large reduction in the phosphorus concentrations of the waters entering the basin. However, while the regeneration rate would remain relatively unchanged for a number of years, the sedimentation rate of phosphorus from the waters would probably diminish with the decreasing concentrations (Table 8). The result of this would be that the net elimination rate of phosphorus would decrease with time. Thus the Western basin waters would not entirely reflect the reduction of phosphorus loadings but, nevertheless, could be expected to show considerable improvement.

Epilimnion Budgets

The phytoplankton biomass during the summer is of the greatest concern in Lake Erie because undue growth at this time of the year makes the lake unpleasant for recreational use and results in the Central basin hypolimnion going anoxic (Burns and Ross 1972). If an investigation of epilimnion budgets discloses the fact that the summer supply of nutrients to the epilimnion is largely due to nutrient resupply from the hypolimnion and not due to external inputs, then it can be projected that the lake will take a long time to respond to corrective action.

In the construction of a materials budget for a well-mixed, unstratified basin, it is not possible to calculate the actual regeneration of material from the sediments. However, this deficiency can be largely overcome in the construction of epilimnion budgets because the supply of materials from the hypolimnion to the epilimnion can be estimated from changes in the volume of hypolimnion and by the use of the Mesolimnion Exchange Model (Burns 1976b).

Soluble Reactive Silica (SRS)

The summer epilimnion budgets for SRS (Fig. 4a) show a significant conversion of soluble

silica to the particulate form in the Central basin (Fig. 4a) but show the reverse process in the Eastern basin (Fig. 4b). Nevertheless, there was an overall increase in the mean epilimnion concentration of SRS from 1.55 to 2.27 μmol liter^{-1} during the summer. An examination of the Central basin SRS epilimnion budget shows that the major source of SRS to the epilimnion was the increase in concentration of 1.58 μmol liter^{-1} resulting from the input of SRS from the hypolimnion (incorporation and exchange). The only significant external input to the Central basin was from the Western basin resulting in an increase in concentration of 0.56 μmol liter^{-1} and did not really constitute an external input to the lake. Thus the Central basin can recycle SRS during the summer at a rate sufficient to maintain a high diatom population (Munawar and Munawar 1976) and the supply of this phytoplankton nutrient is not controllable by man to any significant degree.

The Eastern basin epilimnion budget shows an increase in the concentration of SRS between survey 3 and 7 which cannot be explained in terms of the six basin loading mechanisms investigated (equation 9). An inspection of the silica distribution maps (figure 16, Burns 1976a) shows an increase in SRS concentrations in all areas of the basin, suggesting that there was dissolution of SRS from both the sediments and diatom frustules suspended in the epilimnion. The Eastern basin is different from the Central basin in that comparatively little of the recycled SRS was seen to come from the hypolimnion but that most came from the conversion of particulate silica to the soluble form within the epilimnion. As a result of this there was adequate SRS within the Eastern basin epilimnion for diatom growth, but this did not occur (Munawar and Munawar 1976).

The reason for the large internal loading of silica from the hypolimnion in the Central basin was that the difference in concentration between the epilimnion and hypolimnion was large. Although a considerable quantity of SRS in the Eastern basin was transferred from the hypolimnion to the epilimnion, the net budget hypolimnion SRS inputs were small because the concentration difference between the hypolimnion and epilimnion was small in the Eastern basin.

Soluble Inorganic Nitrogen (SIN)

There were considerable areas in the Central Basin where the concentrations of ammonia and nitrate plus nitrite were less than 1 μm liter^{-1} between surveys 6 and 7 (figures 13 and 14, Burns 1976a), and it is possible that SIN (NH_3 + NO_3^- + NO_2^-) was a growth-limiting nutrient at that time. For this reason, it becomes important to understand the epilimnion budget because if it is possible to reduce the SIN supply to the epilimnion, the resulting nitrogen limitation would affect the phytoplankton species and populations.

The budget (Fig. 4c and d) shows that the largest source of the SIN utilized during the stratified period in both basins was that material which was in the epilimnion at the start of the season (survey 3 concentration). The high initial epilimnion concentration of SIN appears to be due largely to river input to the lake during the period of spring runoff and a reduction of this loading could result in some decrease in the summer availability of SIN. However, reduction of SIN concentrations would not necessarily result in a reduction of phytoplankton concentrations, but could cause a change to increasing concentrations of blue-green species because of their ability to fix atmospheric nitrogen.

In the Eastern basin, the concentration changes resulting from the loadings from the Central basin inflow and Niagara outflow (Fig. 4d) both indicate that the water flowing into and out of the basin had lower concentrations of SIN than the mean concentration of the basin epilimnion (see equation 9). The epilimnion budget shows that the external inputs to the basin would have resulted in a small concentration increase whereas the concentration of SIN was actually decreased greatly by phytoplankton uptake during the summer. It is difficult to be certain what quantity of SIN in the water just prior to stratification was due to external input and how much was due to internal loading within the basin. Because of these factors it is not possible to comment on the controllability of SIN in the Eastern basin. However, these epilimnion budgets demonstrate that in both basins the internal loading of SIN during the summer is small and that the nutrient undergoes major depletion. The concentrations of SIN probably control the phytoplankton growth in the two basins to a significant degree during the summer.

Total Phosphorus

An interesting fact of the phosphorus epilimnion budget data is that in both basins there is a large initial decrease in phosphorus concentration from the epilimnion (Table 9, surveys 3 to 4) which cannot be accounted for by the six mechanisms included in equation (9). This loss diminishes with the length of the stratified period and even becomes a net increase in phosphorus

TABLE 9. Mean epilimnion concentration changes due to phosphorus sedimentation.

Survey interval	Central basin ($\mu mol\ liter^{-1}$)	Eastern basin ($\mu mol\ liter^{-1}$)
3–4	−0.27	−0.18
4–5	−0.20	−0.16
5–6	−0.05	−0.09
6–7	−0.02	+0.02

concentration in the East Basin during the interval between surveys 6 and 7. Since in the case of total phosphorus, we are dealing with the element and not just a specific form of the element, as in the case of SRS and SIN, it is difficult to explain the appearance of a quantity of the element. The disappearance of the phosphorus during the other survey interval periods was presumably due to sedimentation. A possible explanation for the decreasing loss of phosphorus with time is that some of the phosphorus, sedimented out along the shoreline areas where the epilimnion is in direct contact with the sediments, began to redissolve with decay of the organic material. However, this does not seem to be the case because the higher concentrations of phosphorus near the shorelines disappeared rather than appeared, as the stratified season progressed (Fig. 5).

There are two other facts that can be linked to explain the pattern of decreased loss and eventual reappearance of phosphorus in the epilimnia with increased duration of stratification. The diatom species of phytoplankton were replaced by flagellate species to a large extent in the Central basin and almost totally replaced in the Eastern basin during the stratified season (Munawar and Munawar 1976). Secondly, as the season progressed, the concentration difference between soluble reactive phosphorus (SRP) hypolimnion and epilimnion concentrations increased from there being no significant difference in June, to the hypolimnion concentration being 3.4 times greater than the epilimnion concentration in September (Burns et al. 1976). SRP is the most available form of phosphorus to algae and the phytoplankton were probably becoming limited in their growth by lack of SRP as the season progressed because of the elimination of this form of phosphorus from the epilimnion. The flagellates, with their ability of vertical migration (Eppley et al. 1968), could sink into the hypolimnion, absorb the SRP, and migrate upwards again to obtain light for growth. The phytoplankton thus could be a "nutrient pump" acting in the opposite direction to the settling mechanism. Zooplankton migration probably also plays some part in this vertical transport but this requires further study. Vertical transport of phosphorus upward through a thermocline has been observed (Burns and Pashley 1974) so it is considered probable that there was upward vertical transport of phosphorus by phytoplankton through the mesolimnia of both basins.

However, it appears that the Central basin epilimnion does need a continual recharge of phosphorus to replace the material lost from the system by sedimentation since the loading to the epilimnion would have resulted in a concentration increase equal to 79.1% of the concentration decrease caused by the loss of phosphorus from the epilimnion (Fig. 4e). Approximately 47% of the loading to the epilimnion was from rivers and municipalities. Most of the river inputs are from the south shore of the Central basin and tend to be held in that area (Fig. 5). The water quality of the southshore area is worse than that of the north shore and the greatest improvement could be expected in this area if river loads were reduced. A reduction of phosphorus loading to the Western basin would indirectly cause some improvement to the Central basin condition by decreasing the phosphorus content in the waters flowing from the Western to the Central basin. However, since the Western basin input would have caused only a small increase in the Central basin epilimnion phosphorus concentrations, a large Western basin loading reduction would have only a minor effect on the Central basin phosphorus concentrations. Nevertheless, each loading reduction would diminish the extent of the hypolimnetic anoxic conditions and the resulting high phosphorus concentrations in the hypolimnion (Burns and Ross 1972). This would in turn diminish the possibility of the hypolimnion input of phosphorus from increasing the epilimnion concentrations.

Apparently the Central Basin total phosphorus concentrations are maintained at a fairly constant level throughout the summer by the combined external and internal inputs (small net concentration change). A reduction of the external inputs would be a direct cause of small decreased total phosphorus concentrations but would also be an indirect cause of reduced epilimnion concentrations resulting from reduced internal inputs from the hypolimnion. Since the present heavy loadings of fine-grained sediments to the Central basin (Kemp et al. 1976) would tend to bury the present phosphorus-rich sediments, a decrease in the extent of the anoxic conditions would be expected. Thus, if external phosphorus inputs were significantly reduced, the character of the offshore waters of the Central basin would prob-

ably become similar to that of the Eastern basin waters.

The Eastern basin epilimnion phosphorus budget (Fig. 4f) is quite different from that of the Central basin since the majority of the material lost from the epilimnion (85.2%) did not come from external loadings and resulted in decreasing epilimnetic phosphorus concentrations. Also the epilimnion–hypolimnion interactions caused a decrease in the epilimnion phosphorus concentrations because the mean epilimnion total phosphorus concentrations were greater than the mean hypolimnion concentrations. It would seem that a reduction in the summer phosphorus content of the Eastern basin can be brought about only by reducing the concentration of phosphorus in the water at the start of the stratified season. However, the low net removal rate of phosphorus to the sediments (0.18% day^{-1}, Table 8) indicates that there is a great deal of recycling of phosphorus within this water mass and loading reductions may not reduce the prestratification phosphorus concentrations.

Changes in the open waters of the Eastern basin resulting from a decrease in phosphorus loading would be small and very difficult to observe. On account of large-scale recycling, it would probably take many years of reduced loadings to measureably decrease the phosphorus levels in the Eastern basin. However, the southshore areas of the Eastern basin are directly affected by the Central basin southshore loadings and these waters would likely show a response to reduced loadings before the open waters of the basin would do so. Similarly, the northshore areas of the Eastern basin are significantly affected by the phosphorus loadings of the Grand River, Ontario and the shoreline areas in the proximity of this input would probably show a quick response to a decrease in the phosphorus outputs of this river.

Summary and Conclusions

A simple water budget was calculated for Lake Erie for the year 1970 and was found to predict the evaporation and precipitation pattern that is normal for the lake, leading to the conclusion that the calculated water budget was basically correct. Using the concentrations of chloride ion measured in the rivers as well as the river flows (established as correct by the water budget), it was possible to develop a chloride budget for the lake which was within 3.8% of being balanced. The chloride budget was also used to estimate that the 1970 *Martin Karlsen* ship surveys of Lake Erie produced data that were adequate for the estimation of the quantity of a material within the lake with a precision of about 2.6%. Since the river loading data on chloride and phosphorus came from the same sources, it was assumed that the levels of accuracy obtained with the chloride budgets also applied to the phosphorus budgets.

The data on inputs and outputs and on quantities of material within the lake having been found adequate, a whole lake budget for phosphorus was developed. By using interbasin water transports, basin budgets were constructed for total phosphorus. The basin budgets made possible the observation of periods of high phosphorus sedimentation following algal blooms and the onset of stratification. The periods of phosphorus regeneration resulting from anoxic conditions in near bottom water and from the disturbance of sediments due to storms were also observed. In general, the basin budgets resulted in the observation that phosphorus elimination from water was inversely related to water depth and directly related to phosphorus concentration. The high elimination rate of phosphorus from shallow waters is thought to be due to the frequent resuspension of clay materials having a high iron to phosphorus ratio and thus a strong phosphorus adsorption capacity.

The use of the mesolimnion exchange model enabled the estimation of the quantity of nutrients moving up from the hypolimnion into the epilimnion and thus the construction of epilimnion budgets for the Central and Eastern basins. By means of these budgets it was possible to compare the effects of various sources of supply on the epilimnion nutrient concentrations and thus determine the relative magnitudes of the external and internal loadings of a specific nutrient. The investigation revealed that the sources of soluble reactive silica during the summer were almost completely internal, indicating that there is no effective means by which the supply of this nutrient could be changed by man. The epilimnion budgets for soluble inorganic nitrogen revealed that the major source of this nutrient during the summer was the high concentrations existing in the water prior to stratification. These high concentrations were due to both internal loading within the basins and a high external loading during the spring run-off period. Thus, the nitrogen loading to the epilimnion is partially controllable, and some reduction could be achieved by limiting the input of nitrogen compounds in the spring. The phosphorus epilimnion budgets show that the Central basin epilimnion has a small but significant input of phosphorus from external sources during the summer, with the Eastern basin epilimnion obtaining little phos-

phorus from external sources. The Western basin has an extremely high elimination coefficient for phosphorus, and large continuous inputs of this element are required to maintain the phosphorus concentrations at the observed levels.

If phosphorus inputs to the lake are reduced to 14,600 metric tons as planned (U.S. Department of State 1972), it is possible to speculate that the following changes in the water quality of the lake would be observed: the Western basin would respond rapidly with the concentration of phosphorus decreasing with time, and the southshore waters of the Central basin would also show a marked improvement resulting from decreased phosphorus concentrations. The open and northshore water of the Central basin would show a slow improvement in water quality which would commence only some years after the loading reductions had been made, but which could continue for many years. The north and south shoreline areas of the Eastern basin would show improvement within a few years but the open water of the basin could take many years to show measurable change.

Acknowledgments

I thank Dr R. A. Vollenweider and Dr J.-M Jaquet for assistance received during the preparation of this work. I also thank Mr R. A. Winklhofer and Mr N. A. Thomas for providing data collected by the United States Protection Agency on chloride and phosphorus loadings to Lake Erie.

BRITT, N. W. 1955. Stratification in western Lake Erie in the summer of 1953: effects on the *Hexagenia* population. Ecology 36: 239–244.

BURNS, N. M. 1976a. Temperature, oxygen, and nutrient distribution patterns in Lake Erie, 1970. J. Fish. Res. Board Can. 33: 485–511.

1976b. Oxygen depletion in the Central and Eastern basins of Lake Erie during 1970. J. Fish. Res. Board Can. 33: 512–519.

BURNS, N. M., AND J. O. NRIAGU. 1976. Forms of iron and manganese in Lake Erie waters. J. Fish. Res. Board Can. 33: 463–470.

BURNS, N. M., AND A. E. PASHLEY. 1974. In situ measurement of the settling velocity profile of particulate organic carbon in Lake Ontario. J. Fish. Res. Board Can. 31: 291–297.

BURNS, N. M., F. ROSA, AND C. H. CHAN. 1976. Lake Erie water chemistry data 1970–71. Can. Cent. Inland Waters, Burlington, Ont. Pap. 16.

BURNS, N. M., AND C. ROSS. 1972. Oxygen–nutrient relationships within the Central Basin of Lake Erie, p. 85–119. *In* N. M. Burns and C. Ross [ed.] Project hypo. Can. Cent. Inland Waters, Burlington, Ont. Pap. 6.

EPPLEY, R. W., O. HOLM-HANSEN, AND J. D. H. STRICKLAND. 1968. Some observations on the vertical migration of dinoflagellates. J. Phycol. 4: 333–340.

IMBODEN, D. M. 1974. Phosphorus model of lake eutrophication. Limnol. Oceanogr. 19: 297–304.

INTERNATIONAL LAKE ERIE WATER POLLUTION BOARD. 1969. Report to the International Joint Commission on the pollution of Lake Erie, Lake Ontario and the international section of the St. Lawrence River. Vol. 2. Lake Erie.

JOHNSON, M. G., AND G. E. OWEN. 1971. Nutrients and nutrient budgets in the Bay of Quinte, Lake Ontario. J. Water Pollut. Control Fed. 43: 836–853.

KEMP, A. L. W., R. L. THOMAS, C. I. DELL, AND J.-M. JAQUET. 1976. Cultural impact on the geochemistry of sediments in Lake Erie. J. Fish. Res. Board Can. 33: 440–462.

KRAMER, J. R., H. E. ALLEN, G. W. BAULNE, AND N. M. BURNS. 1970. Lake Erie time study. Can. Cent. Inland Waters, Burlington, Ont. Pap. 6.

LAM, D. C. L., AND J.-M. JAQUET. 1976. Computations of physical transport and regeneration of phosphorus in Lake Erie, fall 1970. J. Fish Res. Board Can. 33: 550–563.

LAM, D. C. L., AND T. J. SIMONS. 1976. Numerical computations of advective diffusive transports of chloride in Lake Erie, 1970. J. Fish. Res. Board Can. 33: 537–549.

LEAN, D. R. S. 1973. Phosphorus dynamics in lake Water. Science 179: 678–680.

MARINE SCIENCES DIRECTORATE. 1973. Monthly and yearly mean water levels. Dep. Environ., Ottawa, Ont.

MUNAWAR, M., AND I. F. MUNAWAR. 1976. A lakewide study of phytoplankton biomass and its species composition in Lake Erie, April–December 1970. J. Fish. Res. Board Can. 33: 581–600.

ONGLEY, E. D. 1976. Sediment yields and nutrient loadings from Canadian watersheds tributary to Lake Erie: an overview. J. Fish. Res. Board Can. 33: 471–484.

PHILIPS, D. W., AND J. A. MCCULLOCH. 1972. The climate of the Great Lakes Basin, EN57-7/20. Information Canada, Ottawa, Ont. 40 p.

SIMONS, T. J. 1976. Continuous dynamical calculations of water transports in Lake Erie for 1970. J. Fish. Res. Board Can. 33: 371–384.

UNITED STATES DEPARTMENT OF STATE. 1972. Great Lakes water quality agreement with annexes and texts and terms of reference between the United States of America and Canada. U.S. Government Printing Office, Washington, D.C. T1AS7312.

UNITED STATES GEOLOGICAL SURVEY. 1970, 1967. Water resources data for New York, Albany, N.Y. 12201; Water Resources data for Ohio, Columbus, Ohio 43212; Water resources data for Michigan; Okemos, Mich. 48864.

VOLLENWEIDER, R. A. 1975. Input–output models. Schweiz. Z. Hydrol. 37: 54–84.

1976. Rotsee, a source not a sink for phosphorus. Schweiz. Z. Hydrol. 38. (In press)

WATER SURVEY OF CANADA. 1970. Surface water data for Ontario. Inland Waters Dir., Ottawa, Ont. 190 p.

WILLIAMS, J. D. H., T. P. MURPHY, AND T. MAYER. 1976. Forms of phosphorus in the surficial sediments of Lake Erie. J. Fish. Res. Board Can. 33: 430–439.

Numerical Computations of Advective and Diffusive Transports of Chloride in Lake Erie, 1970[1]

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LAM, D. C. L., AND T. J. SIMONS. 1976. Numerical computations of advective and diffusive transports of chloride in Lake Erie, 1970. J. Fish. Res. Board Can. 33: 537–549.

This paper is concerned with a numerical simulation of lakewide advective and diffusive transports of conservative pollutants or nutrients, such as chloride, in Lake Erie. The study is a continuation of a hydrodynamic modeling experiment to compute water circulations throughout the 1970 shipping season, also in this issue, and makes use of ship observations of chloride distributions at approximately 4-wk intervals during 1970, together with estimates of river loadings for the same periods.

The model equations are based on the time-dependent, mass-conserving advection–diffusion equation. A vertically mixed, one-layer, two-dimensional model is employed for spring and fall, whereas a two-layer model is used to simulate the stratified lake in summer. The equations are solved by finite-difference formulations on a horizontal grid with a mesh size of 6.67 km. Using time steps of 6 h, predictions are made of the change of chloride concentrations between pairs of cruises and the results are compared with observations.

While the advective currents are derived from the previous hydrodynamic study, diffusion coefficients are estimated by comparing model results for a range of coefficients. The resulting numerical values compare favorably with findings of diffusion experiments in actual lake environments.

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Le présent article porte sur une simulation numérique des transports par advection et diffusion à la grandeur du lac de polluants ou d'éléments nutritifs persistants, tels les chlorures, dans le lac Erié. L'étude est un prolongement de l'expérience sur un modèle hydrodynamique construit dans le but de calculer la circulation de l'eau durant toute la saison de navigation de 1970, qui est dans ce numero aussi, est utilise des données cueillies par des navires sur la distribution des chlorures dans le lac Erié à des intervalles d'environ 4 semaines en 1970, de même que des estimés des apports de rivières pour les mêmes périodes.

Les équations du modèle sont fondées sur l'équation advection–diffusion conservatrice de masse et dépendante du temps. On utilise pour le printemps et l'automne un modèle à deux dimensions, à couche unique mélangée verticalement, alors qu'un modèle à deux couches sert à simuler le lac stratifié en été. Les équations sont résolues par formulations de différences finies sur un grillage horizontal de 6.67 km de côté. En utilisant des intervalles de temps de 6 h, nous prédisons les changements de teneur en chlorures entre paires de croisières et comparons les résultats avec les observations.

Alors que les courants advectifs sont calculés à partir de l'étude hydrodynamique antérieure, les coefficients de diffusion sont estimés en comparant les résultats du modèle pour une gamme de coefficients. Les valeurs numériques qui en résultent se comparent favorablement aux résultats d'expériences de diffusion dans des environnements réels du lac.

Accepted November 18, 1975 Accepté le 18 novembre 1975

THE advective and diffusive transports of substances are two physical processes which must be taken into consideration in developing simulation models of the ecosystems of large water bodies. Models that assume horizontal homogeneity in order to study the kinetics of aquatic systems must be expanded to include horizontal water movements if they are to predict the temporal and spatial distributions of the substances in large water bodies such as the Great Lakes (Boyce 1974). The effects of these transport mechanisms, while being equally important for the distribution of nonconservative substances, can be conveni-

[1]Some results of this work have been presented at the 17th Conference on Great Lakes Research, August 12–14, 1974, McMaster University, Hamilton, Ont.

Printed in Canada (J4068)
Imprimé au Canada (J4068)

ently evaluated by modeling the behavior of conservative material.

In recent years, a number of efforts have been made to model horizontal distributions of dissolved or suspended material in the Great Lakes. For instance, Boyce and Hamblin (1975) followed an analytical approach to obtain steady-state distributions of chloride in the Central basin of Lake Erie. The advent of the high-speed computer has actually introduced a much more versatile technique to simulate material transports in natural waters. By dividing the whole lake in a large number of elements, it is possible to compute the time-dependent distribution of any material in the lake, including, effects of winds, shoreline configuration, bottom topography, and arbitrary diffusion parameters. Typical examples of such effects have been illustrated by a simulation of the Niagara River plume in Lake Ontario by Simons (1972).

Numerical computations of the steady-state distributions of chloride in the Western basin of Lake Erie were obtained by Hydroscience Inc. (1973). A segmentation of the basin into 88 arbitrary elements, based on the location of significant gradients of the water quality parameters, constituted the framework for this study. Richardson (1974) used the same method to simulate the long-term averaged distribution of chloride in Saginaw Bay. In these studies, reasonable agreement with observations was obtained for horizontal diffusion coefficients in the range from 10^5 to 10^6 cm^2/s, comparable to experimental values for similar space scales. Sequences of computed steady-state currents, based on 1970 wind data, were used in a time-dependent advection–diffusion model for chloride concentrations in Lake Michigan by Schwab and Katz (1974). Because of lack of data, no verification of this model was presented.

Some of these steady-state models do serve the purpose of describing the general, long-term, time-averaged, transport movements in the water bodies concerned. However, as inherent in any steady-state computation, these long-term mean values tend to smooth out, and hence conceal, some of the more interesting phenomena in cases with large time fluctuations. One example is the transient transport of a large, sudden, river discharge of pollutants; another is the displacement of concentration patterns during a storm. These modes of transport are not infrequently observed in the data, e.g. the chemical parameters observed on Lake Erie during 1970 and discussed at various places in this volume. They certainly do have an important bearing on the understanding of the characteristic advective and diffusive transports of pollutants or nutrients in the water bodies concerned. It is, therefore, desirable that a continuous, time-dependent advection–diffusion transport model be developed to cope with both the general and the extreme cases, using actual data on winds, bathemetry, lake boundaries, thermocline structures, and river sources.

It is based on such rationale that the present model is designed. The purposes here are twofold. First, it will be used for a comprehensive simulation of the basic processes involved in large scale advection–diffusion and thus provide reasonable estimates of the diffusion coefficients by a prognostic approach. Second, it will be verified extensively against observations so that one can achieve an adequate level of confidence to adapt it for modeling nonconservative substances, nutrient availability, and other basic biochemical processes. Also included in this issue is a mathematical model for phosphorous processes as sequel to the present study (Lam and Jaquet 1976).

The currents required to simulate advective processes in the lake are obtained from a continuous hydrodynamical modeling study of Lake Erie for the same period (Simons 1976). The results of the latter investigation are stored in the form of daily averaged currents on magnetic tapes and thus are directly accessible to the present model. Initial estimates of diffusion parameters are derived from numerous determinations of dispersion processes in the Great Lakes by Murthy (1971) and Murthy et al. (1974). To verify the present model, a series of episodic observations of chloride distributions is needed. In this regard, the 10 CCIW monitor cruises on Lake Erie in 1970 provide an excellent data base.

1. Mathematical Model

In the context of this special issue of papers concentrating on the biochemical constituents of Lake Erie, the discussion of the mathematical model will be more qualitative than quantitative. To facilitate the discussion, a general form of the model equations will be presented but there appears no need to list the numerical formulations actually used for the calculations. Some technical details will be presented in the next section, under the title, "Numerical Considerations," which may be omitted without affecting the continuity of the discussion.

In this study a vertically mixed model is employed for the spring and fall seasons, whereas a two-layer configuration is adopted for the summer stratified lake with the thermocline serving as

the separating interface. Both models are prognostic in character such that the variables, in this case the chloride concentrations, are computed forward in time from specified initial values. Thus the input parameters are the bottom topography and shoreline configuration, the thermocline depths, the previously computed currents, the diffusion coefficients, the initial values of chloride concentration, and the river loadings. The output parameters are the final chloride concentrations at a specified later time and possibly accumulative statistical quantities pertaining to the advection and/or diffusion of chloride across specified sections of the lake. The equations are solved numerically by superimposing on the lake, a regular grid with a mesh size of 6.67 km and by predicting the changes for each cell in a sequence of time steps of 6 h each.

The model will be based on the time-dependent advection–diffusion equation, which is essentially a statement of mass conservation. The total lake is divided into volume elements and the principle of mass conservation dictates that the time rate of change of material contained in each cell must be equal to the sum of: 1) the so-called advective transport of material through the sides, top, and bottom of the volume elements by more-or-less organized water motions; 2) the dispersive transport through the same boundaries by irregular water displacements, usually called sub-grid-scale diffusion to indicate that the scale of these motions is less than the mesh-size of the grid; and, 3) external sources and sinks such as river loadings or chemical reactions resulting in an increase of one substance at the expense of another.

Consider the mass balance for an individual cell of the grid. If the mesh size of the horizontal grid is denoted by d, the local depth of a model layer by h, and the mean concentration of the volume element by c, then the mass contained in this element is d^2hc. The transport of material through any side of the grid cell is equal to $dhvc$, where v is the component of the velocity perpendicular to this side, and the transport through the top or bottom equals d^2wc, where w is the vertical velocity component. In the latter transport formulations, c is the concentration of the volume element under consideration or an adjacent cell or an appropriate combination of these two, depending on the direction of the flow through the respective boundary. The time rate of change of the mass contained in the element equals the sum of the transports through the six faces of our grid element plus the external sources. This is our basic model equation expressed in non-mathematical terms. The crucial point, however, is that we must define more precisely what we mean by transports.

The velocities and concentrations in the model equations are actually mean values over a certain length and time scale. In our present testing case, the velocities are computed from a grid resolution of 6.67 km and averaged over a daily interval, and hence, they represent mean transports with a length scale of 6.67 km. Since we use an interpolation scheme to define the concentration at all our grid cells and assume homogeneity within each cell, the concentration can be also considered a mean value with length scale 6.67 km. Here, our emphasis on the length scale arises from the fact that the diffusion coefficients depend on the length scale considered. During the derivation of our model equations, we could have written the equations in terms of the original turbulent variables, but these variables would have been too oscillatory in both time and space and would have rendered our computations impossible. Thus, we introduce a mean component and a fluctuation component to each of these variables, e.g. $v = \bar{v} + v'$ and $c = \bar{c} + c'$. We are then able to bring our equations into a more manageable form, except that the equations are no longer in closed form, because the product terms give rise to terms like $\overline{v'c'}$. Many hypotheses have been proposed to bring these turbulent fluctuation terms into closed form. The most commonly used one is the gradient hypothesis which states that the turbulent exchange is proportional to the concentration gradient; thus, $\overline{v'c'} = -\mathrm{A}\ \partial c/\partial n$, where A is an effective diffusion coefficient and n indicates the direction normal to the surface through which the exchange takes place (Okubo 1971). We will return to this subject in the discussion of the results of our study.

2. Numerical Considerations

The governing equations for the present model are time-dependent advection–diffusion equations as discussed qualitatively in the previous section. Integrated over a layer of water of depth h contained between two arbitrary surfaces, these equations take on the general form:

$$\frac{\partial}{\partial t}(hc) = -\nabla\cdot(\vec{v}hc) + \nabla\cdot(\mathrm{A}h\nabla c) + (wc)_2 - (wc)_1 + \left(\mathrm{K}\frac{\partial c}{\partial z}\right)_1 - \left(\mathrm{K}\frac{\partial c}{\partial z}\right)_2 + \mathrm{S}$$

where t is time, ∇ is the horizontal gradient operator, z is the vertical coordinate, $\vec{v}$ is the horizontal velocity vector, w is the vertical water

displacement perpendicular to the bounding surfaces at top and bottom (denoted by 1, and 2, respectively), c is the concentration of the parameter of interest, A and K are horizontal and vertical eddy diffusion coefficients, and S represents all external sources. A derivation of the equation in this particular form has been presented by Simons (1973), including a discussion of asumptions and simplifications. Note that the depth h is in general a function of both time and horizontal coordinates.

For the vertically mixed lake model, the right-hand side of the equation reduces to the first two terms plus the source term. For the two-layer summer model, vertical advective and diffusive transports occur only at the thermocline, i.e. at the bottom of the top layer and the top of the bottom layer. For the length scale of the present grid, i.e. 6.67 km, experiments by Murthy et al. (1974) indicate horizontal diffusion coefficients in the epilimnion of 2.5×10^5 cm^2/s, and for the hypolimnion, Murthy (1971) obtained values ranging from 10^3 to 10^4 cm^2/s. Since the thermocline separates the layers in the summer model, the vertical diffusion coefficient should be comparatively small. Murthy et al. (1974) suggest values of K in the range from 0.1 to 0.5 cm^2/s. All horizontal and vertical velocities are obtained from a hydrodynamic model (Simons 1976).

The finite-difference leapfrog scheme (Browning et al. 1973; Richtmeyer and Morton 1967) is used to solve the model equations numerically. The scheme is based on the same spatial grid as used in the hydrodynamic model (Simons 1976), i.e. 60×16 square elements with $\Delta x = \Delta y =$ 6.67 km. A time step of up to 6 h can be used without yielding unstable results. The details of the numerical techniques, stability analysis, computational and storage efficiency of the present scheme will not be discussed in this paper. We shall only briefly explain here some of the salient features of the computational aspects.

Let us consider again the two-layer model. We shall apply the equation above to each of the grid cells. Each column of water may consist of two elementary rectangular volumes separated by the thermocline or only one elementary volume if there is no thermocline. Our strategy is such that we place the horizontal transport velocities at the midpoints of the sides of the corresponding elementary volumes, while defining the mean concentrations at the center of the upper volume and at the center of the lower volume. The vertical velocity will be at the center of the interfacing plane. The equations are then discretized over the usual five-point spatial rectangular grid and three time levels, so that the scheme is explicit but achieves second order accuracy in the time derivative, with the advection terms evaluated at the intermediate time level and the diffusion terms at the older time level. The discretization is taken over a total of 596 cells in the lake.

There are some computationally interesting problems that require special treatment in order to obtain reasonable solutions. One problem is that for regions with high velocities, the advection–diffusion equations tend to be more hyperbolic so that central difference discretization of the advection terms may not be appropriate for low resolution grids (Torrance 1968; Price et al. 1966; Lam 1973). On the other hand, sudden release of high concentrations of pollutants from the river to the lake may give rise to computational singularities. This further complicates the problem of the general treatment of boundary conditions. We use a mass-conservative version of upstreaming differencing for the advective terms and time smoothing techniques to deal with these problems.

3. Applications

To test our model, we apply it to the CCIW chloride data collected in Lake Erie during the shipping season of 1970. There are 56 observation stations and we use an objective interpolation procedure (Simons 1976) to define the value of chloride concentration at all the 596 grid points in the lake. We also take the chloride inputs from 21 rivers which flow into or out of Lake Erie for the same period. Since the time intervals between the cruises for which the chloride observations are available vary from 17 to 61 days, we must group the river inputs proportionally to correspond to these intervals on a time-varying basis. There are 10 cruises (or surveys) in 1970, with chloride observed in all but the second of them. Nevertheless, we shall label these 10 cruises in the original chronological order.

Our procedure is to use the data of cruise 1 as the initial condition and proceed to solve our model equation numerically forward in time until we reach the time of the next cruise observation, i.e. cruise 3 because cruise 2 chloride data are missing. We then compare the chloride distribution predicted by our computation with the observed data. Next, we start again, using the observed data of cruise 3 as our initial condition and proceed to cruise 4 and compare the predicted and the observed chloride distributions. This is carried further from cruise 4 to 5, and then cruise 5 to 6 and so on. The purpose of this verification exercise is to elucidate, by means of

the model, major changes in the observed chloride due to advection and diffusion.

Three particular aspects of the computations require some further discussion. The first problem concerns the river loadings of chloride. Some of the river inputs during the period of interest were actually not available and had to be estimated from previous records. The most significan one is the Grand River, Ohio, which contributes a substantial amount of the total chloride inputs. Also, the distribution of chloride concentrations across the mouth of a large river, such as the Detroit River, can have a significant effect on the chloride distributions in adjacent lake waters. As an example, Fig. 1 shows the results of two simulations for the same amount of total chloride loadings from the Detroit River, but in the upper figure all chloride is carried by the eastern half of the river while in the lower figure the chloride is contained in the western half. Measurement of the actual outflow across the river mouth shows that the first model is closer to the truth. In our calculations, we assumed that the outflow from the eastern half of the river was twice as large as that from the western half. Notice that the computed plumes are very similar to those observed by Hartley et al. (1966).

The next problem concerns the horizontal diffusion processes which were briefly touched upon in section 1. As mentioned before, the length scale corresponding to our grid mesh is $L = 6.67 \times 10^5$ cm. Experimental results (Murthy et al. 1974) suggest that the diffusion coefficient, A, is related to the length scale as $A = 5.6 \times 10^{-3} L^{1.3}$ so that A should be approximately 2.5×10^5 cm^2/s for the epilimnion in our model. We have tried different values for the epilimnion horizontal diffusion coefficient varying from 10^4 to 10^6 cm^2/s. It is found that while we can use 7.5×10^4 to 2.5×10^5 cm^2/s to obtain reasonable results, higher values (say 10^6 cm^2/s) tend to diffuse concentration very quickly and lower values (say 10^4 cm/s^2) tend to cause unrealistically high gradients. As an example, Fig. 2 shows computed results for different values of A as compared to the observed distribution for two cruises in spring. In the same way, the working range for the horizontal diffusion coefficient in the hypolimnion is found to be 10^3–10^4 cm^2/s and the vertical diffusion coefficient across the thermocline should be very small, i.e. of the order 0.1 cm^2/s. All these values of diffusion coefficients agree with experimental findings (e.g. Murthy et al. 1974). However, the working range of these coefficients determined from our verification procedure is quite wide because the concentration gradients of chloride in the lake are

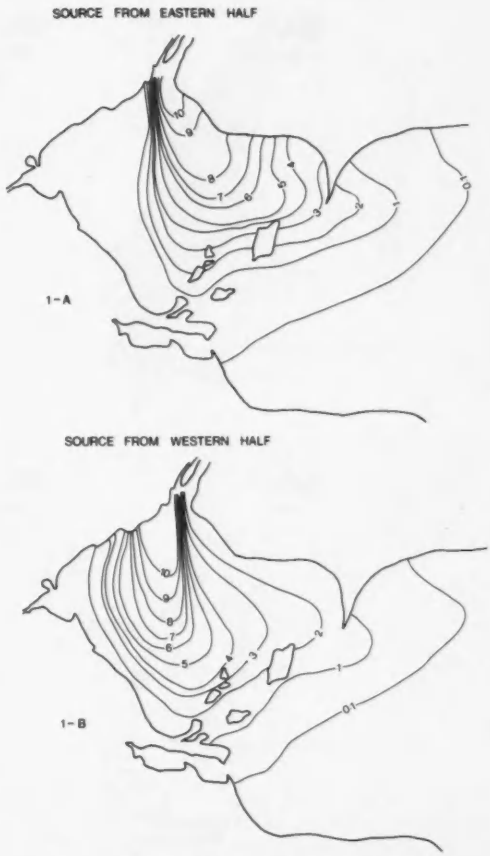

FIG. 1. Effect of chloride concentration gradient across the mouth of the Detroit River on the chloride distribution in the Western basin of Lake Erie. The numbers denote the chloride concentrations relative to the strength of the source input. 1-A, All chloride contained in the eastern half of the river. 1-B, All chloride contained in the western half of the river.

generally too small to work out a finer range of values. Yet, it indicates that the formulas obtained from observations are applicable in modeling studies. To demonstrate the effects of advection and diffusion for the length scale in our model, Fig. 3 shows the depth-averaged, computed advective and diffusive fluxes across the boundary sections between the basins at different times of the year.

A third problem is related to the vertical exchange of material at the interface of the two-layer summer model. Since this model interface supposedly represents the thermocline in the real

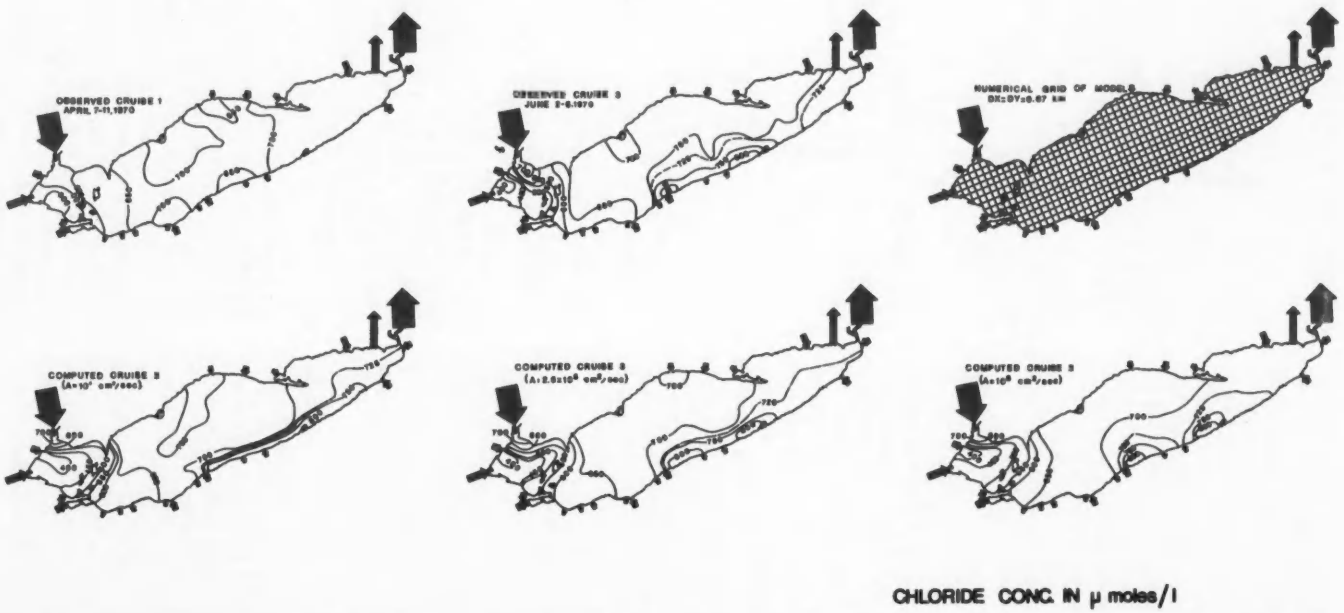

FIG. 2. Observed vs. computed chloride concentrations for indicated dates during the spring of 1970. Parameter A represents value of horizontal diffusion coefficient used in computations.

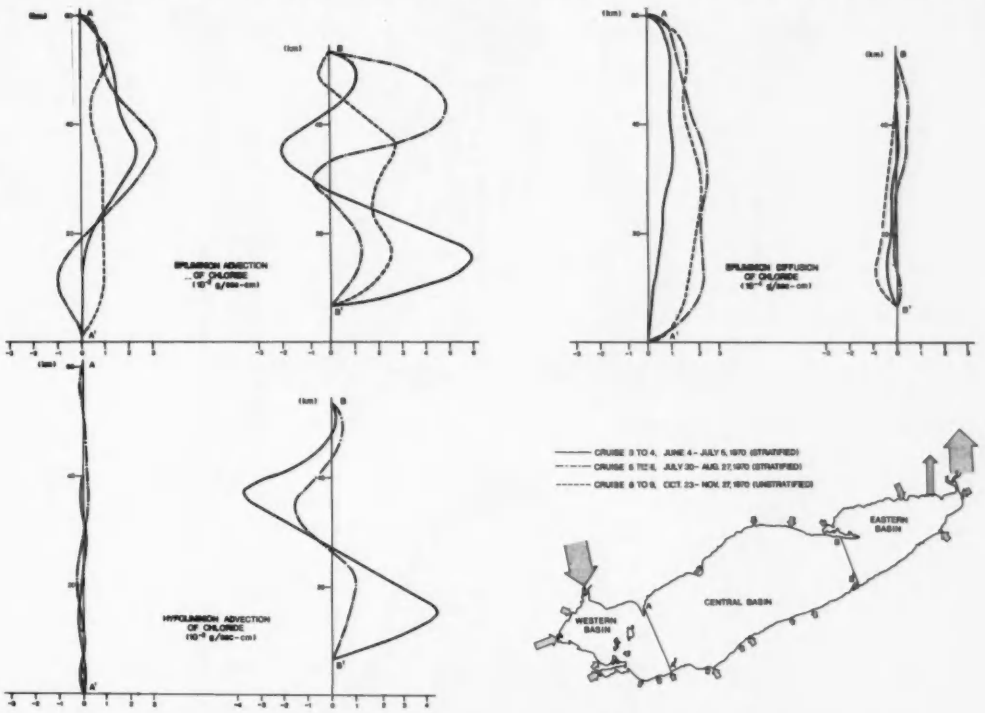
EPILIMNION ADVECTION OF CHLORIDE
EPILIMNION DIFFUSION OF CHLORIDE
HYPOLIMNION ADVECTION OF CHLORIDE
CRUISE 3 TO 4, JUNE 4 – JULY 5, 1970 (STRATIFIED)
CRUISE 5 TO 6, JULY 30 – AUG. 27, 1970 (STRATIFIED)
CRUISE 8 TO 9, OCT. 23 – NOV. 27, 1970 (UNSTRATIFIED)
EASTERN BASIN
CENTRAL BASIN
WESTERN BASIN

FIG. 3. Computed advective and diffusive fluxes per unit length across the sections AA′ and BB′.

lake, it is clear that it should act as an effective barrier against exchanges between epilimnion and hypolimnion. On the other hand, it is known that the thermocline position may vary over longer periods of time, such as considered here, which results in apparent mixing of epilimnion and hypolimnion waters. On a shorter time scale, important mixing processes take place in the form of upwelling and downwelling induced by wind action. These processes are mostly confined to narrow zones near the shores and are difficult to simulate with models of the present resolution. This question of vertical transports actually belongs to the realm of the hydrodynamic model, since the vertical velocities at the interface or the interface displacements are directly coupled to the currents in the model layers. As discussed in the paper on the latter subject (Simons 1976), the present two-layer hydrodynamic model attempted to solve the problem by using observed thermocline positions derived from the 1970 Lake Erie cruises. As a result, any thermocline deflections in the nearshore zones caused by upwelling or downwelling show up in the model as vertical water transports across the interface, and consequently, represent mixing of the two layers. This naturally overestimates the actual exchanges due to wind-induced vertical circulations. However, as mentioned in the hydrodynamic model paper, there is a simple technique to separate the total currents computed by the model such that this mixing effect can be eliminated or reduced by any amount desired. To evaluate this effect all two-layer computations of chloride concentrations were carried out in two ways. In the first case, the mean vertical velocities were computed for each cruise interval and then subtracted from the daily values. This precludes an exchange of water over the long term but allows for exchanges during stormy periods. It obviously still allows for a net long-term exchange of chloride between epilimnion and hypolimnion because the transport is the product of vertical velocity and chloride concentration, and hence, we are dealing with a nonlinear process. In the second case, the vertical velocities were used as computed originally by the hydrodynamic model. As an example, Fig. 4 shows the results of both calculations for the interval between cruises 3 and 4. The first method appears to give better

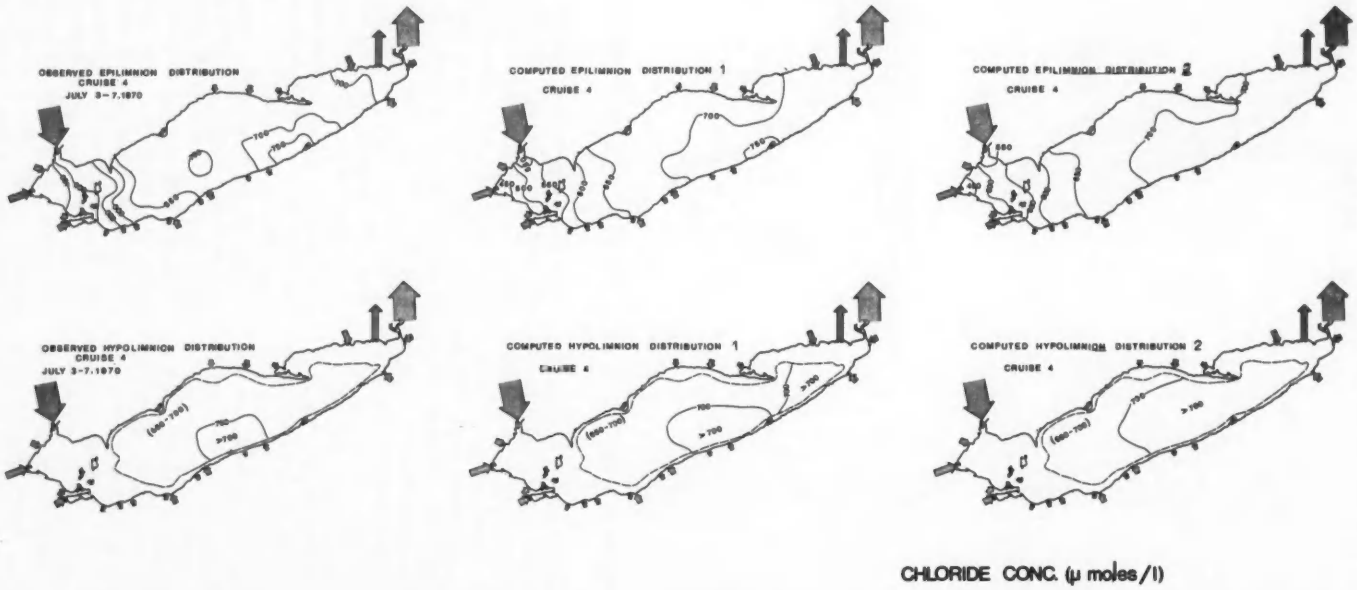

FIG. 4. Observed and computed chloride concentrations in epilimnion and hypolimnion for indicated dates. Computations were based on methods (1) and (2) as described in the text.

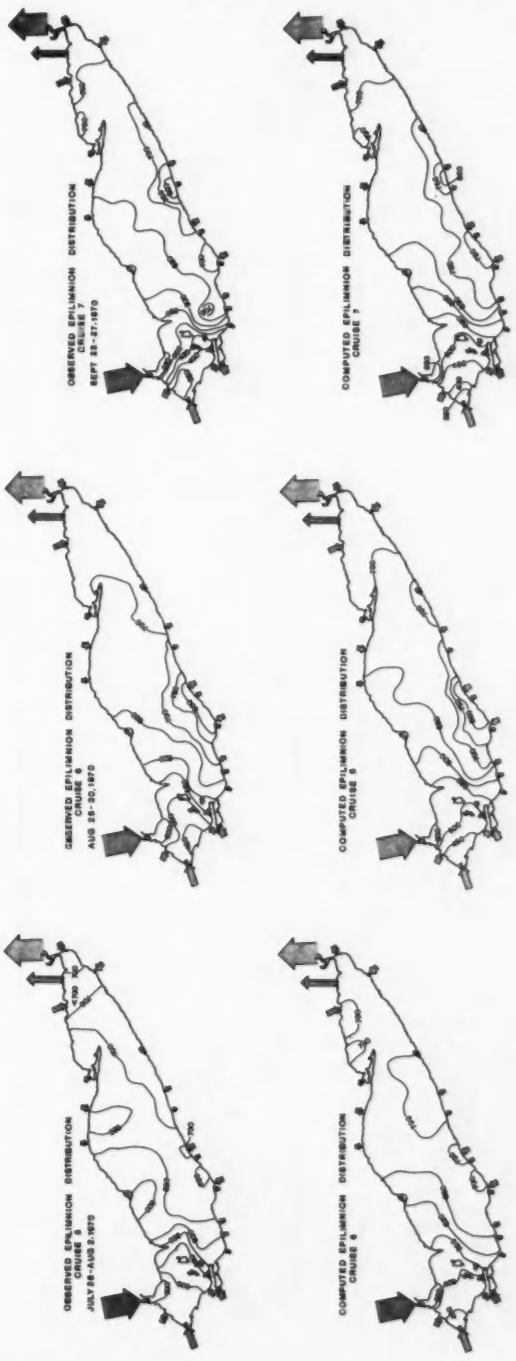

FIG. 5. Observed and computed chloride concentrations in epilimnion for indicated cruises during the summer of 1970.

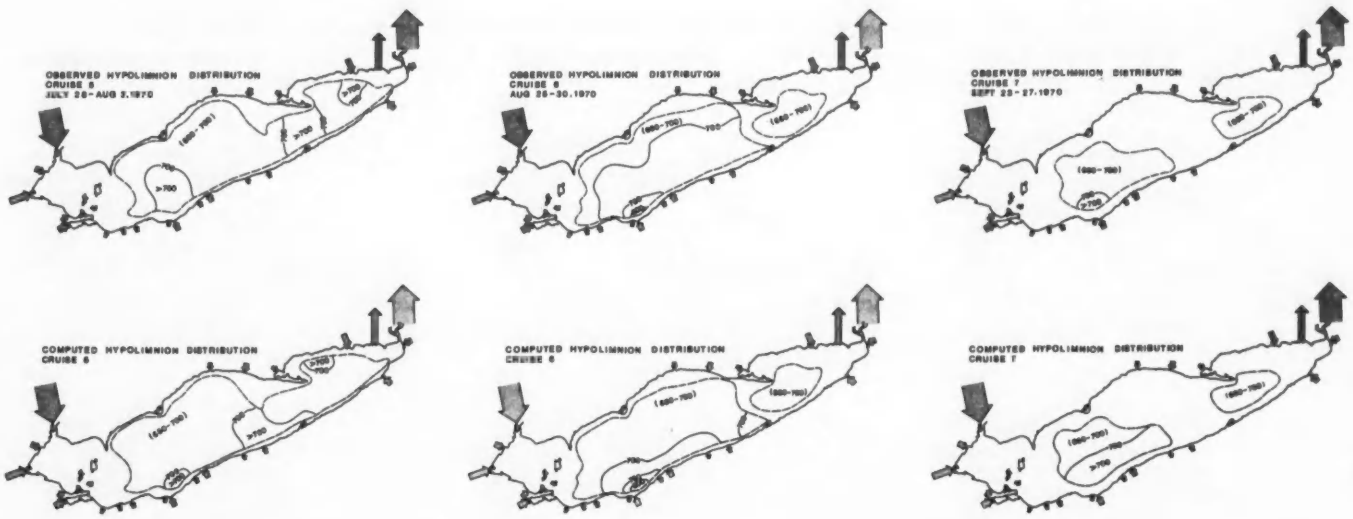

FIG. 6. Observed and computed chloride concentrations in hypolimnion for indicated cruises during the summer of 1970.

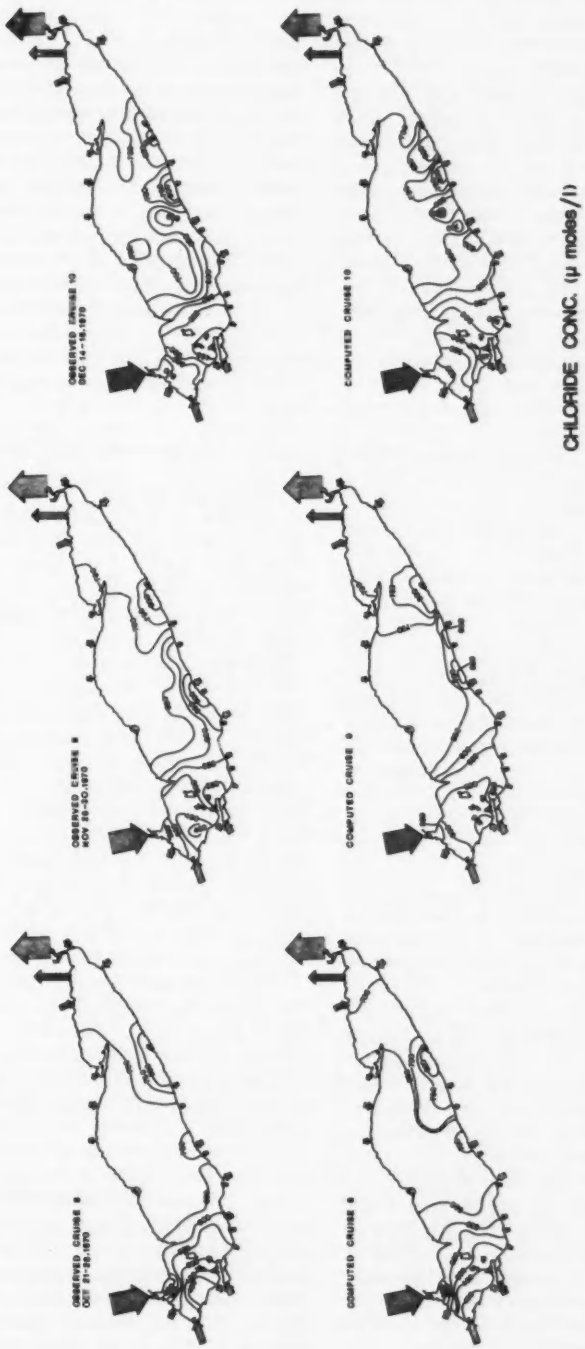

FIG. 7. Observed and computed chloride concentrations during the fall of 1970.

agreement with observations and thus has been adopted for the calculations for which the results are presented in the following figures.

In Fig. 5, we show the computed and observed chloride distributions in the epilimnion from cruise 4 to cruise 7, each time using the previous cruise data as our initial values for the prediction. In Fig. 6, we display similar comparison maps for the hypolimnion. Both results have been obtained from the two-layer model. Finally, Fig. 7 shows the results for the fall season obtained from the vertically mixed model. For all the computations, the results were checked for mass conservation, i.e. the total quality of chloride in the lake at the beginning of the computation is equal to the quantity at the end of the computation plus the addition or subtraction due to river sources or sinks.

Several advective and diffusive transport characteristics have been noticed during the model verification exercise. When going from cruise 1 to cruise 3, (Fig. 2), we obtain an overall agreement with the observed distribution; in particular, we observe high concentrations of chloride due to local river discharges on the south shore of the Central basin. The gradient of chloride concentration in the Eastern basin is always small because it is a well-mixed region. When we go from cruise 3 to cruise 4 (Fig. 4), the two patches, that initially formed on the south shore, are carried by the nearshore currents into the Eastern basin, leaving only one high concentration zone (the 750 μmol/liter-contour) with a trailing lower concentration patch (650 μmol/liter-contour). This is revealed both in the computed and observed maps.

Note that the shape of the 650 μmol/liter-contour that roughly separates the Western and the Central basin remains more or less the same for cruise 3 and cruise 4. However, when we compute the distribution for cruise 5, (Fig. 5), based on the cruise 4 data as initial values, this particular 650 μmol/liter-contour is twisted around. This is due to a clockwise circulation which appears to develop in the Central basin during summer stratification (Simons 1976; Fig. 4). This result is confirmed by the observed distribution map for cruise 5. This particular contour holds more or less the same shape for the next two cruise intervals, i.e. cruise 5 to 6 and cruise 6 to 7, until the start of fall turnover. It swings back to its original shape as observed in cruises 3 and 4, when we assume a vertical mixing of the hypolimnion and the epilimnion for the computation from cruise 7 to 8, (Fig. 7). This again agrees with the observed distribution.

Meanwhile, a patch of high concentration (800 μmol/liter-contour), which originates near the Grand River, Ohio, on the south shore of Lake Erie during cruise 6, moves steadily towards the Eastern basin. Its diffusive and advective movements can be traced in both the computed and the observed distribution maps for cruises 6, 7, and 8. The period from cruise 8 to cruise 9 sees little chloride inputs from the rivers, but a storm develops towards the end of the cruise interval, resulting in large water displacements. Finally, we have the shortest cruise interval (17 days) from cruise 9 to 10, where high winds still dominate and the currents often change their directions. All this is reflected in an intermingling of high and low concentration patches in the Central basin of the Lake, as shown in the computed distribution map as well as the observed map.

4. Summary and Conclusions

This study attempted to simulate the transport of chloride in Lake Erie throughout the 1970 shipping season. Within the framework of available observations on chloride distributions at monthly intervals and estimates of river loadings, the changes of chloride concentrations were computed from one month to the next by numerical techniques. The simulation model is based on the time-dependent advection–diffusion equation which is solved on a horizontal grid with a mesh size of 6.67 km and with time steps of 6 h. During the spring and fall seasons, the lake is represented by a vertically mixed model, whereas the summer lake is simulated by a two-layer configuration. The currents required for these computations were computed previously by hydrodynamic models of similar structure, while diffusion coefficients were estimated from observations as well as from a comparison of model results for a range of coefficients.

Among the problems treated in some detail are the effect of variations in chloride discharges across the mouth of the Detroit River, the relation of effective diffusion coefficients to grid size and temperature stratification, and the vertical transport of material across the thermocline. It is found that a diffusion coefficient of 2.5×10^5 cm^2/s produces satisfactory results for the epilimnion. This value is in agreement with observed diffusion processes for spatial scales corresponding to the present grid mesh. In regard to vertical exchange of chloride at the thermocline caused by upwelling and downwelling, the computations suggest that the monthly-mean exchange of water between epilimnion and hypolimnion is likely to be smaller than computed by the particular two-layer model adopted for the hydrodynamic study. Therefore, an approxima-

tion is used (Simons 1976) to eliminate the monthly-mean vertical velocity at the interface without eliminating daiiy exchanges due to windy periods. Since the exchange of chloride is related to correlations between instantaneous velocities and chloride concentrations, this essentially limits such exchanges to periods of significant upwelling, but it does not prevent a net exchange of chloride over the full cruise interval.

Recent efforts have been directed towards establishing an objective criterion for comparing computed and observed distributions. Such a criterion should take the form of a skill test or score such as the conventional performance test applied to numerical weather-prediction models. Although preliminary tests of similar criteria show reasonable correlations between computed and observed distributions, the question of statistical significance must be further evaluated. Furthermore, it should be pointed out that, among a number of simulations of a particular episode, the one simulation with the best objective score is not necessarily the most appealing to an observer. In effect, a subjective comparison tends to single out certain auspicious phenomena, such as a patch of high concentration moving along the shore. This kind of verification is difficult to include in an objective verification criterion.

The results presented in this paper have been chosen from a number of simulations for a range of parameters on the basis of subjective evaluation by the authors. Within the restrictions of this procedure, the performance of the model employed in the present study must be judged satisfactory. The figures included in this paper show that several characteristic advective and diffusive transport phenomena in large lakes can be depicted by such models. The same models can be readily applied to nonconservative water quality parameters and will play an essential role in simulating transports of nutrients and biochemical parameters in the aquatic environment. This transport framework will make it possible to isolate biochemical processes and their relations with regard to the total ecosystems of large lakes or seas.

Acknowledgments

This research was carried out while D. C. L. Lam was supported by a National Research Council of Canada Postdoctorate Fellowship. The chloride data were processed and analyzed by N. M. Burns and C. H. Chan.

BOYCE, F. M. 1974. Some aspects of Great Lakes physics of importance to biological and chemical processes. J. Fish. Res. Board Can. 31: 689–730.

BOYCE, F. M., AND P. F. HAMBLIN. 1975. A simple diffusion model of the meanfield distribution of soluble materials in the Great Lakes. Limnol. Oceanogr. 20: (In press)

BROWNING, F., H. O. KREISS, AND J. OLIGER. 1973. Mesh refinement. Math. Comp. 27: 29–39.

HARTLEY, R. P., C. E. HERDENDORF, AND M. KELLER. 1966. Synoptic water sampling survey of the Western Basin of Lake Erie. Proc. Conf. Great Lakes Res. 9: 301–322.

HYDROSCIENCE, INC. 1973. Limnological systems analysis of the Great Lakes, Phase 1 — Preliminary model design. Hydroscience Inc., Westwood, New Jersey. 474 p.

LAM, D. C. L. 1973. Modal analysis of semidiscrete problems for the diffusion–convection equation. Proc. Man. Conf. Numer. Math. 1973, Univ. Manitoba, Winnipeg, Man. 3: 277–286.

LAM, D. C. L., AND J. M. JAQUET. 1976. Computations of physical transport and regeneration of phosphorus in Lake Erie, fall 1970. J. Fish. Res. Board Can. 33: 550–563.

MURTHY, C. R. 1971. An investigation of diffusion characteristics of the hypolimnion of Lake Erie. Proc. Conf. Great Lakes Res. 14: 799–804.

MURTHY, C. R., G. KULLENBERG, H. WESTERBERG, AND K. C. MINERS. 1974. Large scale diffusion studies. Can. Cent. Inland Waters Pap. 14: 19 p.

OKUBO, A. 1971. Horizontal and vertical mixing in the sea. p. 89–168. *In* D. W. Hood [ed.] Impingement of man on the oceans. John Wiley and Sons, Inc., New York, N.Y.

PRICE, H. S., R. S. VARGA, AND J. E. WARREN. 1966. Application of oscillation matrices to diffusion–convection equations. J. Math. Phys. (Cambridge, Mass.) 45: 301–311.

RICHARDSON, W. L. 1974. Modelling chloride distribution in Saginaw Bay. Proc. Conf. Great Lakes Res. 17: 462–470.

RICHTMEYER, R. D., AND K. W. MORTON. 1967. Difference methods for initial-value problems. Interscience Publishers, New York, N.Y.

SCHWAB, G. M., AND P. L. KATZ. 1974. A model for the study of episodes in the dispersion of a conservative pollutant in Lake Michigan. Proc. Conf. Great Lakes Res. 17: 837–845.

SIMONS, T. J. 1972. Development of numerical models of Lake Ontario, Part 2. Proc. Conf. Great Lakes Res. 15: 655–672.

1973. Development of three-dimensional numerical models of the Great Lakes. Can. Cent. Inland Waters Sci. Ser. 12: 26.

1976. Continuous dynamical computations of water transports in Lake Erie for 1970. J. Fish. Res. Board Can. 33: 371–384.

TORRANCE, K. E. 1968. Comparison of finite-difference computations of natural convection. J. Res. N.B.S.—Math. Sci. 72B: 281–301.

Computations of Physical Transport and Regeneration of Phosphorus in Lake Erie, Fall 1970[1]

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LAM, D. C. L., AND J.-M. JAQUET. 1976. Computations of physical transport and regeneration of phosphorus in Lake Erie, fall 1970. J. Fish. Res. Board Can. 33: 550–563.

A one-layer, two-dimensional computer model for the time-dependent, lakewide advective and diffusive transports as well as physical regeneration of total phosphorus in Lake Erie has been developed. It has 60 × 16 square grid cells of 6.67-km mesh size and uses a time step of 12 h. The model has been verified and found to be in reasonably good agreement with the Canada Centre for Inland Waters (CCIW) monitor cruise data for October, November, and December during which period there were high winds with high phosphorus concentrations being observed. Computed, daily averaged currents from a hydrodynamic model developed at CCIW are used for the transport terms and actual data for the lake boundary conditions. The physical regeneration is attributed to wave motions induced by wind. A formula is proposed which expresses the regenerated amount of total phosphorus in terms of wave orbital velocity and sediment mean grain size. A discussion is presented on the choice of the settling rate and the regeneration coefficient which produce satisfactory model results.

LAM, D. C. L., AND J.-M. JAQUET. 1976. Computations of physical transport and regeneration of phosphorus in Lake Erie, fall 1970. J. Fish. Res. Board Can. 33: 550–563.

Les auteurs ont mis au point un modèle d'ordinateur à deux dimensions et à couche unique pour les transports par advection et par diffusion dans tout le lac dépendant du temps, de même que pour la régénération physique du phosphore total dans le lac Erié. Le modèle est constitué d'un grillage de 60 × 16 cellules carrés de 6.67 km de côté et utilise un intervalle de temps de 12 h. Il a été vérifié et jugé en assez bon accord avec les données cueillies lors de croisières de routine du CCEI en octobre, novembre et décembre, période au cours de laquelle les vents ont été forts et de hautes teneurs en phosphore ont été observées. Les courants calculés et les moyennes quotidiennes établies à partir d'un modèle hydrodynamique développé au CCEI (Simons 1976) servent à définir les termes de transport et les données réelles se rapportant à la couche limite du lac. On attribue aux mouvements des vagues causées par le vent la régénération physique. On suggère une formule exprimant la quantité de phosphore total régénéré en termes de vélocité orbitale des vagues et de grosseur moyenne des particules du sédiment. On discute du choix du taux de sédimentation et du coefficient de régénération produisant des résultats satisfaisants pour le modèle.

Accepted November 18, 1975

Accepté le 18 novembre 1975

THE phosphorus cycle in large lakes is complex. Even in the simplest terms, one must distinguish between processes involving the soluble form of phosphorus and those involving the insoluble form, or particulate phosphorus. Of fundamental importance are: 1) the advective and diffusive processes which are responsible primarily for the horizontal movement of both soluble and particulate phosphorus; 2) the phenomenon of downward settling of particulate phosphorus; and 3) the upward regeneration of soluble and particulate phosphorus from the sediments due to biological, chemical, or physical causes. Superimposed on these individual mechanisms are the conversion of phosphorus from one form to the other and the interrelations of each form with other water quality parameters.

This paper is a first attempt at studying synthetically some of these basic processes by means of computer simulation. More specifically, a model is presented which simulates the time-dependent, horizontal advective and diffusive transports as well as physical settling and regeneration of phosphorus under high wind and nonstratified conditions in Lake Erie.

Thus, our model is basically different from the box model of phosphorus proposed by Imboden

[1]Some of the results in this paper were presented at the 18th Conference on Great Lakes Research, SUNY at Albany, N.Y., May 20–23, 1975.

Printed in Canada (J4069)
Imprimé au Canada (J4069)

(1974). We use a higher resolution in time and incorporate the lateral transport in space. This additional degree of freedom of space enables one to predict both the time evolution and spatial variations of phosphorus concentrations in the lake. As a result, the model is able to respond to such effects as the time-varying phosphorus loadings from different river sites, the spatial variation of grain size and the wave-turbulent motions leading to resuspension of bottom sediments. In this way we can bring into the model valuable information from various branches of water science, such as hydrodynamics and sedimentology.

In fact, the present study can be viewed as an endeavor to unify some of the elements brought up in the other papers presented in this issue, particularly in the areas of physical, chemical, and geological aspects of phosphorus in Lake Erie. Thus, the phosphorus data collected during the 1970 CCIW cruises (Burns 1976a, b) were used to verify the model. The computed currents, stored on magnetic tape in the form of daily averaged transports, were taken from the continuous hydrodynamic model (Simons 1976) to compute the advective transports. The computer program of the chloride model (Lam and Simons 1976) was adapted for nonconservative substances such as phosphorus and the range of horizontal diffusion coefficients, as confirmed in the chloride model verification, was used in the phosphorus model. Other processed data and correlation between parameters as presented by Thomas et al. (1976), Williams et al. (1976), and Kemp et al (1976) were also included in the present model. Existing experimental and verified results in the literature were also used to decrease the number of undetermined coefficients in the model equations and formulae.

A general description of the model is given below, followed by a discussion concentrating on the physical aspect of phosphorus regeneration resulting from wave motions. A hypothetical submodel relating such regeneration to wave and sediment characteristics is proposed. Finally, verified results of the model are presented.

The Mathematical Model

The model is based on the time-dependent advection–diffusion equation plus source terms. At the present stage, it is designed for nonstratified conditions since it applies mostly to high wind periods that occur in Lake Erie during and after the autumn overturn. It is predictive in character so that the variables are computed forward in time from specified initial values of spatial distributions. The output variables considered at the moment are the concentrations of total phosphorus [TP] and particulate phosphorus [PP]. The inputs to the model are lake bathymetry, shoreline configuration, initial values of the phosphorus concentrations in lake water, computed currents, river sources and sinks, atmospheric loadings, municipal and industrial discharges, wind speeds and directions, and sediment [TP] content computed from the mean grain size distribution. The outputs include the phosphorus concentrations at a specified later time and place or at any time during the computation, and cumulative statistical values such as the estimated amount of phosphorus regenerated or settled in any area of the lake. It is also possible to compute the advective and diffusive transports of phosphorus across certain sections of the lake over a specified time interval during the computation (Fig. 1).

The governing equations for the phosphorus model are based on the principles of mass conservation:

$$\partial h[TP]/\partial t = -\partial(u[TP])/\partial x - \partial(v[TP])/\partial y + \partial(Ah\partial[TP]/\partial x)/\partial x + \partial(Ah\partial[TP]/\partial y)/\partial y - \sigma[PP] + G + S_{[TP]} \quad (1)$$

$$\partial h[PP]/\partial t = -\partial(u[PP])/\partial x - \partial(v[PP])/\partial y + \partial(Ah\partial[PP]/\partial x)/\partial x + \partial(Ah\partial[PP]/\partial y)/\partial y - \sigma[PP] + \gamma G + S_{[PP]} + \alpha([TP] - [PP]) - \beta[PP] \quad (2)$$

where x, y = spatial variables (cm)
t = time (s)
h = depth (cm)
[TP] = concentration of total phosphorus (g/cm^3)
[PP] = concentration of particulate phosphorus (g/cm^3)
u, v = vertically integrated velocities over the entire depth (cm^2/s)
A = horizontal diffusion coefficient (cm^2/s)
σ = settling rate (cm/s)
G = regeneration flux (g/s cm^2)
γ = fraction of G that pertains to particulate phosphorus
$S_{[TP]}$ = TP source term from rivers, etc. (g/s cm^2)
$S_{[PP]}$ = PP source term from rivers, etc. (g/s cm^2)
α = photosynthesis or production rate (cm/s)
β = respiration or mineralization rate (cm/s)

Considering equation (1), the term on the left-hand side denotes the change of mean concentration of total phosphorus over the whole depth (g/s cm^2). The first and second terms on the right-hand side of the equation are the horizontal advective fluxes (g/s cm^2) in x and y directions. The third and fourth terms are the horizontal diffusive fluxes. The gradient hypothesis is used as in the chloride model (Lam and Simons 1976) and A is taken as 2.5×10^5 cm^2/s. The settling velocity σ in the

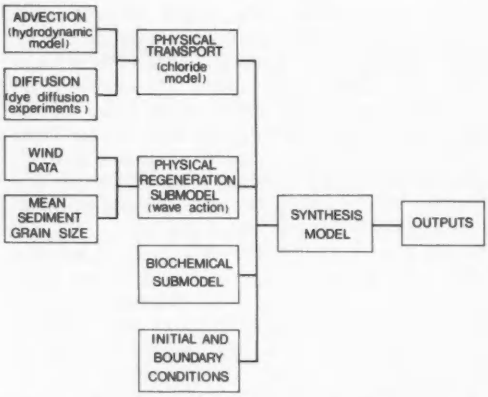


FIG. 1. Components of Lake Erie phosphorus model.

fifth term should be in the range of 0.1–0.4 m/day (Imboden 1974; Burns et al. 1974). The sixth term, G, is the regeneration flux of phosphorus from the sediment. Only the case where G is of physical origin is considered here. The last term in equation (1) refers to the input fluxes from the rivers, municipalities, industries, and the atmosphere.

With regard to equation (2), the terms are similar to those in equation (1) up to the fifth one on the right-hand side. The sixth term, γG, represents the amount of particulate phosphorus regenerated from the sediment. The seventh term, $S_{[PP]}$, corresponds to $S_{[TP]}$. The eighth term, $+\alpha([TP] - [PP])$, is the conversion term of soluble phosphorus to particulate phosphorus due to photosynthesis. The last term, $-\beta[PP]$, is the conversion term from the particulate phosphorus to soluble phosphorus due to respiration or mineralization.

Let us suppose that the ratio $r = [PP]/[TP]$ remains fairly constant over a large area, e.g. over a lake basin during the time of computation (see later for the case of Lake Erie). Then equation (1) may, in fact, be written in terms of [TP] only:

$$\partial h[TP]/\partial t = -\partial(u[TP])/\partial x - \partial(v[TP])/\partial y + \partial(Ah\partial[TP]/\partial x)/\partial x + \partial(Ah\partial[TP]/\partial y)/\partial y - \sigma r[TP] + G + S_{[TP]} \quad (3)$$

In this case, there are only two undetermined coefficients (i.e. σ and another one in G). This is a great simplification over solving the system of equations (1) and (2) with five possible undetermined coefficients: α, β, γ, σ, and the one in G. Indeed, equation (3) was chosen for the computations and its usage appears justified from the results.

As in the chloride model, the finite difference leapfrog scheme was used to solve the model equations numerically. For example, to solve equation (3), the same spatial grid as used in the hydrodynamic model (Simons 1976) and in the chloride model was taken as the spatial mesh in the present model, i.e. 60×16 square grids with $\Delta x = \Delta y = 6.67$ km. For settling velocities in the range of 0.1–0.6 m/day, a time step of up to $\Delta t = 12$ h can be used without making the computations unstable. The discretization of the equation was then taken over a total of 596 cells in the lake. The source terms and the settling and regeneration terms were evaluated at the intermediate time level, with second order accuracy of Δt. An upstream differencing method for the advective terms and a time smoothing technique were applied to treat problems of strong source or resuspension, particularly at the boundaries of the lake (Lam and Simons 1976). The computed outputs of the model were compared to observed data and will be discussed in a later section on application and verification.

It must be stressed again that the present study aims only at simulating the physical regeneration of total phosphorus. In Lake Erie, for example, there were two periods of phosphorus regeneration during 1970 (Burns 1976a). One period occurred in late summer when the anoxic hypolimnion condition led to phosphorus release by biochemical processes. The other period occurred in the fall when the lake was under high wind conditions; a far greater amount of phosphorus was then regenerated by wave action. Thus, the physical regeneration is important, although it has less deteriorative effects than the biochemical regeneration as it occurs usually in the fall and the physically regenerated phosphorus consists of a large fraction of particulate materials which quickly resettle. Its importance as a significant regeneration mechanism is explained by Moore and Silver (1973) and Moore et al. (1974 and unpublished data). The following description demonstrates how the present model can be used as a learning tool for studying this physical aspect (Fig. 2).

Physical Regeneration

Sediment movement and resuspension can result either from nonperiodic currents, such as those computed by Simons (1976), or from periodic movement of the water at the bottom caused by surface waves. Only the latter process will be considered here as being capable of causing sediment resuspension for the following reasons:

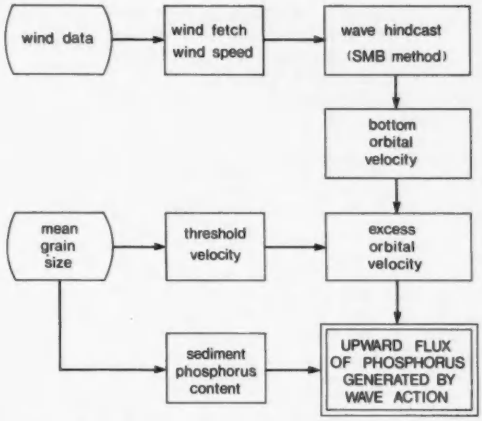


FIG. 2. Structure of physical regeneration submodel.

1) The general current pattern and velocities for Lake Erie have been computed from water transport; velocities are expressed as depth-averaged values or, in other words, they are assumed to have a uniform vertical distribution. This assumption makes the computation of bottom shear stress from current velocities a difficult task.

2) For practically all the periods considered, bottom orbital velocities were about one order of magnitude higher than computed current velocities. Moreover, it is known that an oscillating flow field produces more stress than a constant one (Komar et al. 1973; Yalin et al. 1966; McCave 1973; Migniot 1968). The contribution of the currents may, therefore, be neglected as a first approximation.

The following steps were taken in this part of the model dealing with physical regeneration of phosphorus: starting with wind speed and direction, relevant wave parameters were obtained from a wave hindcasting method and used to compute bottom orbital velocities; sediment resuspension was then expressed as a function of this velocity; and finally, phosphorus regeneration was estimated by taking into account its distribution in sediments (Fig. 2).

WAVE HINDCASTING

Wave hindcasting was performed following the Sverdrup–Munk–Bretschneider (SMB) method as described by King (1972) and U.S. Coastal Engineering Research Center (1973), and programmed by Simpson and Walters (1970 and unpublished data). This semiempirical method provides an estimate of significant wave height and significant wave period at a given location when wind speed, fetch length, and duration are given. In addition, wave length is computed taking water depth into account.

Due to the relatively small size of Lake Erie when compared to oceans, wave attenuation or decay was not considered significant and, therefore, was not included in the hindcasting method.

Wave parameters computed by such a method must be taken with caution. Although the SMB method is, to date, the best one for the Great Lakes, correlations between estimated and measured values are still low (Cole and Ayers 1967). On the other hand, as the hindcasting computer program was originally designed for deepwater conditions, wave height and period in shallow waters are likely to be overestimated. Provision for this effect will be made in an updated version of the program.

WAVE ORBITAL VELOCITIES AT THE BOTTOM

Under the assumptions of small-amplitude wave theory (U.S. Coastal Engineering Research Center 1973), water particles move in circular paths in deep water and in elliptical paths in shallow and transitional water. The maximum horizontal velocity caused by this periodic movement, at the top of the boundary layer (Komar and Miller 1973), is given by

$$u_m = \frac{\pi H_s}{T_s \sinh\left(\frac{2\pi d}{L_d}\right)} \tag{4}$$

in which H_s is the significant wave height (centimeters), T_s the significant wave period (seconds), d the depth of the bottom (centimeters), L_d the wave length (centimeters) for depth d, and u_m the maximum horizontal velocity (centimeters per second). The wave length is given by

$$L_d = L \tanh\left(\frac{2\pi d}{L_d}\right) \tag{5}$$

with

$$L = \frac{gT_s^2}{2\pi} \tag{6}$$

where L is the wave length in deep water ($d/L \geqslant 0.5$) and g is the acceleration due to gravity.

The periodic motion causes a shear stress τ_o to develop at the sediment–water interface. When τ_o reaches a certain critical value τ_{cr}, the top layer of the sediment is set into motion and may eventually be resuspended at higher values to τ_o. If shear velocity u_* is used instead of τ_o, with

$$\tau_o = \rho u_*^2 \tag{7}$$

where ρ is the density of water, then the relation between u_m and u_* is given by Komar and Miller (1973) as

$$u_* = Cu_m \tag{8}$$

with

$$C = (0.5\, C_f)^{1/2} \tag{9}$$

where C_f is a dimensionless drag or friction coefficient. Inman (1957) assumed that C is a constant but Riedel and Kamphuis (1973) have shown that for oscillatory flow, C_f depends on flow conditions and bottom roughness. Using the graphical method of these authors for prevailing conditions in Lake Erie (T_s = 2–5 *s*, u_m = 2–100 cm/s), C_f may be shown to vary between 0.103 and 0.003. Substituting these values in equations (9) and (8) yields the approximate relation between u_* and u_m presented in Table 1.

TABLE 1. Relation between orbital velocity u_m and shear velocity u_* for extreme wave periods [from equations (8) and (9)].

	u_* (cm/s)	
u_m (cm/s)	T = 2 (s)	T = 5 (s)
2	0.5	0.4
3	0.6	0.5
5	0.8	0.6
10	1.1	0.9
20	1.5	1.1
50	2.4	2.2
100	4.5	4.2

SEDIMENT RESUSPENSION

According to the data available for Lake Erie as well as other water bodies (Williams and Mayer 1972; J. D. H. Williams unpublished data), most of the phosphorus which can be effectively regenerated is contained in the fine-grained sediments. In Lake Erie, they occur in basins with water depth between 15 and 40 m and consist of clay and silty clay with a water content of about 90% at the interface. Organic carbon varies between 1 and 6% (Kemp et al. 1976).

Fine sediments exhibit cohesive properties increasing with clay content, the limit between cohesive ond noncohesive materials lying around a mean particle diameter of 5 phi (30–40 μm). Critical erosion velocities for cohesive sediments depend on many parameters such as grain size and shape, type of clay materials, organic content, water content, etc. (Partheniades 1971; Nielson 1973; Migniot 1968). This last author has shown that for various types of cohesive sediments, the critical shear velocity may be adequately expressed as an inverse function of water content. These findings have been confirmed by Southard et al. (1971) and Lonsdale and Southard (1974).

Recent data on threshold and critical shear velocities are presented in Table 2. They pertain mostly to nonperiodic flow, as information is still meager on critical velocities of cohesive sediments under wave action. The last column of Table 2 shows threshold velocities for bedload movement. Values for suspended load should be higher.

Drawing from the data presented in Tables 1 and 2, critical shear velocities and corresponding threshold orbital velocities were chosen for Lake Erie sediments. They are given in Table 3 for the three size-classes: sand, silt, and clay.

The amount of resuspended sediment is easier to compute for nonperiodic flows (Partheniades 1971; Graf 1971; Lebon and Lambremont 1975) than for periodic flows (Das 1972; Migniot 1968; Hom-ma and Horikawa 1962). As a first approximation in the case of a lakewide phenomenon, the upward flux of sediment was empirically expressed by

$$J = f(u_m, u_{cr}, \rho, \rho_s) \tag{10}$$

where J is the upward flux of resuspended sediment(g/s cm^2) and ρ_s the dry sediment density (2.6 g/cm^3). After a few trials with the model, the following empirical expression was selected as being the most suitable:

$$J = K_r \rho \left(\frac{\rho_s}{\rho_s - \rho}\right) AU \tag{11}$$

This expression bears some resemblance to the bedload mass transport of Sternberg (1972). K_r is a dimensionless proportionality constant equal to 6.4×10^{-14}, $A = u_a/u_{cr}$, u_a being a reference velocity equal to 1 cm/s and $U = u_m - u_{cr}$, the excess orbital velocity (centimeters per second). The sediment flux is, therefore, directly proportional to the excess orbital velocity and inversely proportional to the threshold velocity u_{cr}.

PHOSPHORUS REGENERATION

The various forms of occurrence of phosphorus in Lake Erie sediments are given by Williams et al. (1976) as apatite phosphorus, nonapatite inorganic phosphorus (NAI-P), and organic phosphorus. These authors also explain the relations

TABLE 2. Critical shear velocities and threshold velocities (in brackets) of natural sediment (in centimeters per second).

Source			Migniot 1968	Lonsdale 1974	Southard 1971	Sternberg 1972	Komar 1975
Type of sediment			River muds	Deep-sea clay	Calcareous ooze	Sand	Sand
Cohesive			Yes	Yes	Partly	No	No
Type of flow			Nonperiodic	Nonperiodic	Nonperiodic	Nonperiodic	Periodic
Mean grain size (phi units and μm)	0	1000				3.4[62][b]	[25–30][c]
	1	500				2.5[46]	[15–20]
	2	250				2.0[37]	[10–15]
	3	125				2.3[42]	[7–12]
	4	63				3.0[55]	[5–8]
	5	32					[4–6]
	6	16					[4–5]
	7	8			1.4[15][a]		
	8	4					
	9	2	0.3–1.0				
	10	1	[7–24][a]	1.3[18][a]			
H_2O (% total sed.)			~90	~80	50	Not pertinent	
Organic matter (%)			0.2–4.7	—	—	—	—

[a]Mean velocity in flume.
[b]Mean velocity 1 m above sea floor.
[c]Maximum orbital velocity (top of boundary layer) producing initiation of bedload movement, for periods T = 1–5 s.

TABLE 3. Critical shear velocities and corresponding threshold velocities for Lake Erie surficial sediments.

Size-class		Type of sediment	Water content top 0.5 cm (%)	Threshold velocity (cm/s)	Critical shear velocity (cm/s)
ϕ	μm				
		Sand			
0–4	1000–63	Noncohesive	—	20	1.1–1.5
		Silt			
4–8	63–4	Partly cohesive	90–95	3	0.5–0.6
		Clay			
>8	<4	Strongly cohesive	90–100	2	0.4–0.5

between phosphorus forms and sedimentary parameters. When sediment resuspension is taking place, NAI-P and organic P will contribute to most of the total P analyzed in the overlying water, because these forms are known to be adsorbed on or associated with clay and organic particles. A small amount of soluble phosphorus (IWP) in the order of 0.1 μg/g (Nriagu and Dell 1974) coming from sediment interstitial water will also be regenerated. Apatite should not remain in suspension for a very long time because of its relatively high density (3.1) and coarser grain size, as explained by Williams et al. (1976). When it is resuspended under rough weather conditions, it will remain localized in nearshore waters, where it was not sampled by the 1970 cruises. For these reasons, apatite was not taken into consideration in the regeneration processes.

The amount of phosphorus prone to be regenerated into the water is, therefore, the sum of the NAI-P and organic P fractions of the sediment, IWP being included in NAI-P. It is a function of the mean grain size, as shown in Fig. 3, drawn from data by Williams et al. (1976). The curvilinear trend has been approximated by three linear segments, as given by the following equation:

$$\hat{P}_s = 200 \qquad \bar{\phi} \leq 5.5 \tag{12a}$$

$$\hat{P}_s = 108\bar{\phi} - 394 \qquad 5.5 < \bar{\phi} \leq 7.5 \tag{12b}$$

$$\hat{P}_s = 269\bar{\phi} - 1600 \qquad 7.5 < \bar{\phi} \tag{12c}$$

where $\hat{P}_s$ is the estimate of regeneration-prone phosphorus (micrograms per gram) and $\bar{\phi}$ the sediment mean grain size (phi units).

Finally, the upward flux of regenerated phos-

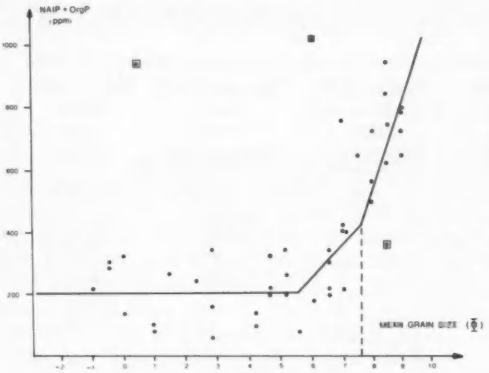


FIG. 3. Relationship of $\hat{P}_s$ (nonapatite inorganic phosphorus, NAI-P, and organic phosphorus, ORG-P) with mean grain size $\bar{\phi}$, in the sediments of Lake Erie (samples marked by squares are anomalous).

phorus (G) is computed by combining (11) and (12) into

$$G = J\hat{P}_s \qquad (13)$$

where G is given in micrograms per cm^2 second.

SETTLING PROCESSES

Acting in the opposite direction to the regeneration processes is settling or downward movement of particulate phosphorus. In the model, we used a constant settling velocity σ in the equations (1) to (3). This simplification was adopted because the settling processes of phosphorus are still poorly understood. Thus, we have essentially two unknown coefficients to be adjusted in the model: σ for settling and K_r for regeneration.

It appears that certain combinations of values for σ and K_r, when applied to the model, will produce more or less the same overall results. However, the choice of these values is not totally arbitrary: whereas the regeneration coefficient K_r is a complete unknown number, σ may be approximated on the basis of settling velocities of particulate matter in lake water (Burns et al. 1974). Moreover, as there is no obvious physical reason why values of σ and K_r should vary with time or location, they were kept constant throughout the computations.

Application

The phosphorus data of the 1970 CCIW cruises 8, 9, and 10 in Lake Erie, as analyzed by Burns (1976a), provide excellent materials for testing our model. During the time interval from cruise 8 to cruise 10 (October 21–December 18), winds were intermittently high, as shown in Fig. 4 (T. J. Simons personal communication), and bottom orbital velocities attained mean values up to 20 cm/s (Fig. 5). Due to a limitation in the program computing the wind fetch, the actual shape of the Long Point Spit could not be represented and the velocities to the northeast of this land feature have been left blank. Large increases of regenerated phosphorus in various parts of the lake were observed in cruises 9 and 10 (see Fig. 6). Indeed, from these facts, the increase can be attributed mainly to physical regeneration.

Moreover, between adjacent cruises for this period, the averaged ratio r = [PP]:[TP] in each of the three basins remained fairly constant (except possibly for the Eastern basin in cruises 9 and 10). They are shown in Table 4 (Burns 1976b) and provide some justification in using equation (3) for testing the proposed physical regeneration submodel. r must be known if equation (3) is used or else γ must be known if the system of equations (1) and (2) is used. Equation (3) was chosen for the sake of simplicity and to focus on total phosphorus regeneration alone. In the computation for the prediction of phosphorus concentrations between cruises, the ratio r is taken as determined from the initial condition of [TP] and [PP] in the basin. Of course, for a more precise simulation, equations (1) and (2) should be used and the information about the ratio would not be needed.

The actual river, municipal, industrial, and atmospheric loadings of total phosphorus were entered as inputs and adjusted for the time spans between the cruises. The averaged wave orbital velocities of 12-h intervals, for each of the 596 points in the lake over the same time spans, were computed from actual 3-h wind records. This involves the use of computer programs for determining effective wind fetch and hence the wave characteristics necessary for the computation of bottom orbital velocities. As in the case of computed currents, these velocities were stored in computer files. Meanwhile, other information such as bathymetry, shoreline, initial TP distribution, and mean grain-size distribution were supplied to the computer model via punched cards.

Verification

During the course of development of the physical regeneration submodel, a series of computational experiments were designed in which the results exerted feedbacks on subsequent trials for improving the etsimate of the regeneration term, G, in equation (3). The idea here was to use the observed [TP] distribution of cruise 8 as the

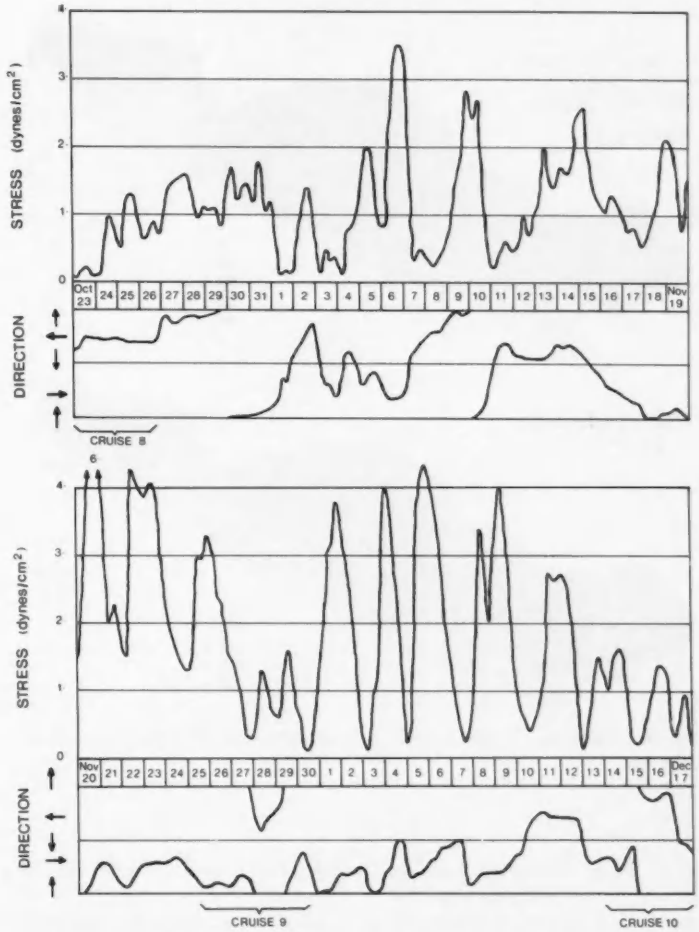


FIG. 4. Wind records averaged over six stations, October 23–December 17, 1970 (T. J. Simons personal communication).

initial condition and then solve the partial differential equation (3) numerically. For each of the 596 cells, the values of [TP] were advanced in time steps of 12 h during a 35-day period until, in this case, the cruise 9 data could be used for verification. During the computation, the time-varying inputs to the model were updated, including the computed currents, wave orbital velocities, and river sources. The computed cruise 9 distribution of [TP] was then compared to the observed cruise 9 distribution. Finally, the simulation experiment was repeated once more by going from the observed cruise 9 distribution as initial condition to a computed cruise 10 distribution, 20 days later. From a series of such simulation results, we determined the choice of the coefficients σ and K_r. We are not able to conclude that, among the alternatives tested, the formula given for (11) is the best or the final one, but only that it is the most satisfactory one so far. Figure 6 shows the total phosphorus distributions for cruise 9 and cruise 10, respectively. Visual comparison of observed and computed distributions show excellent resemblance, especially in the regeneration patterns for the Central basin. A settling rate of $\sigma = 0.25$ m/day and regeneration flux coefficient of $K_r = 6.4 \times 10^{-14}$ were used in the model for all the computations discussed here. A more precise verification of the model by means of statistical techniques such as

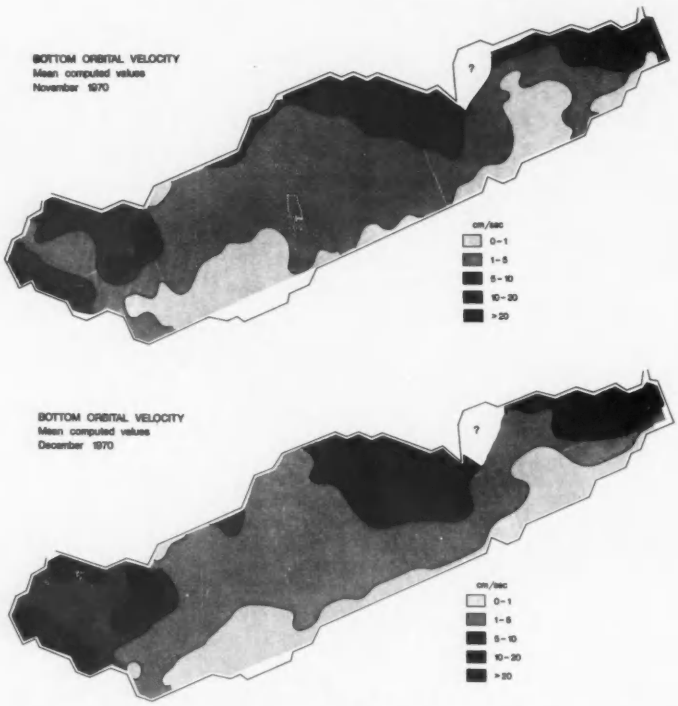


FIG. 5. Computed mean bottom orbital velocities for November and December 1970.

residuals analysis is being conducted and the results will be reported in a subsequent publication. A further illustration of the modeling results is shown in Fig. 7a and b which illustrate the areal distributions of regenerated and settled phosphorus for cruise intervals 8 to 9 and 9 to 10, as computed by the model. These amounts are cumulative sums of the regenerated and settled phosphorus for the particular cruise interval. They are part of the output features of the computer program and were obtained by isolating the other effects such as advection–diffusion at each time step. As mentioned earlier, the settling velocity σ and the regeneration coefficient K_r were chosen in a heuristic manner, so that the regenerated and settled amounts of phosphorus shown separately in Fig. 7a and b may be somehow arbitrary. However, these two amounts, when added together, (bottom maps of Fig. 7a and b) should be a reasonable simulation of the quantity of phosphorus transported vertically, because these values were a major component in producing the predicted P distribution which were then compared with the observed [TP] distributions.

In general, one can conclude from Fig. 7a and b that the spatial patterns of regeneration and settling are quite similar for both cruise intervals, and are largely due to the similarity of geological and meteorological inputs for the two periods.

To provide some idea of the magnitude of the different mechanisms involved, we have also calculated the total cumulative amounts of net regenerated and settled phosphorus in each of the three basins and the total amounts of phosphorus transported across the basin boundaries. The re-

TABLE 4. Ratio of particulate phosphorus [PP] to total phosphorus [TP] by basin and by cruise.

Cruise	Western basin	Central basin	Eastern basin
8	0.66	0.58	0.43
9	0.65	0.60	0.51
10	0.67	0.67	0.58

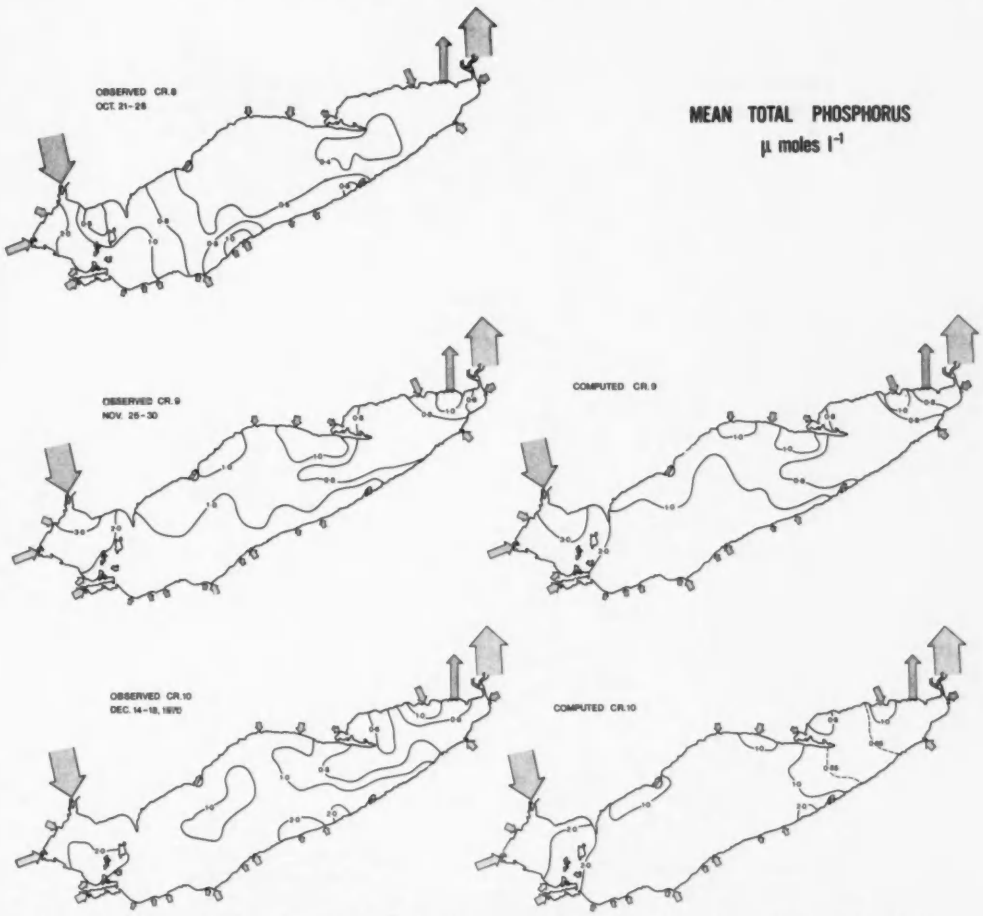


FIG. 6. Observed and computed total phosphorus distribution for November and December 1970.

TABLE 5. Computed total amounts of phosphorus due to regeneration, settling and advection–diffusion (metric tons). Positive sign denotes quantities transported from below or from left-hand side.

	Western basin	Western basin–Central basin		Central basin	Central basin–Eastern basin		Eastern basin
	Net regeneration and settling	Advection	Diffusion	Net regeneration and settling	Advection	Diffusion	Net regeneration and settling
Cruise 8–9	−328	540	6	2774	244	4	537
Cruise 9–10	−2	429	3	2322	343	8	395

sults are shown in Table 5. Considering the much smaller areas of the vertical cross sections between the basins as compared to the areas of the individual basins themselves, the horizontal advective transport fluxes appear to be the most important mechanism. However, in terms of the total amounts involved, the regeneration and settling processes during stormy periods do play an important role in the balance of phosphorus content in Lake Erie.

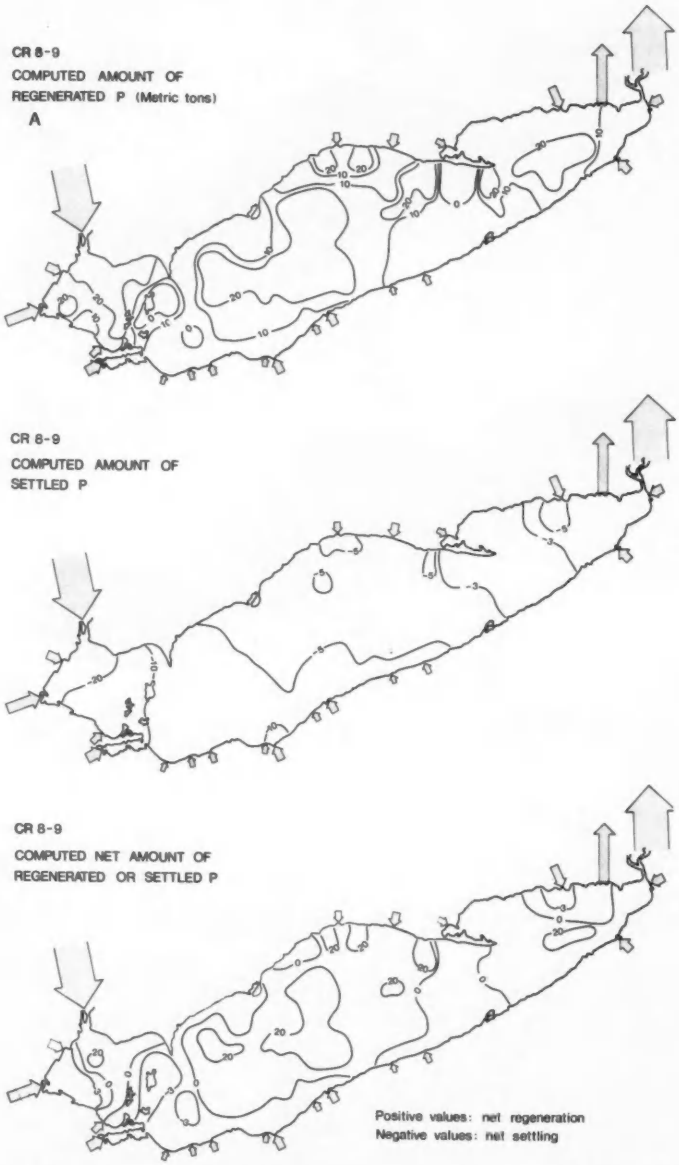
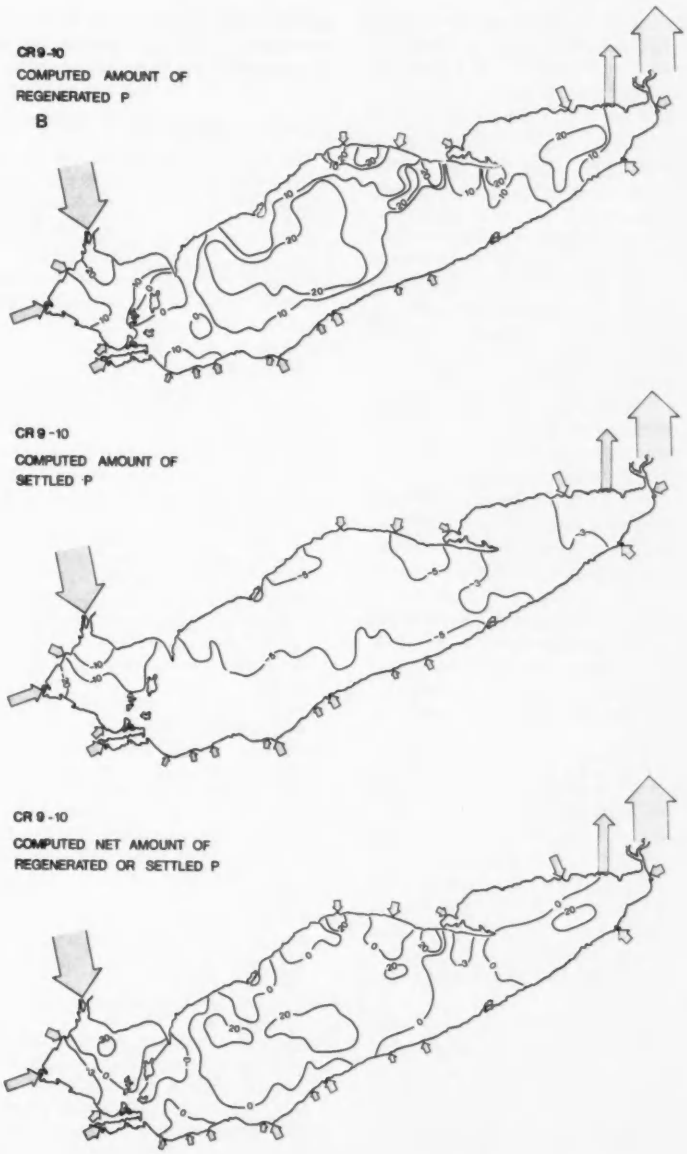


FIG. 7. Computed distribution of regenerated and settled phosphorus for a) cruise intervals 8 to 9 and b) cruise intervals 9 to 10. All values in metric tons. Positive sign denotes addition of phosphorus to lake water from bottom sediments.



Summary and Conclusions

The present study was directed towards the understanding, by computer simulation, of both the physical transport and physical regeneration aspects of phosphorus processes under high wind and unstratified temperature conditions in Lake Erie. A one-layer, two-dimensional, time-dependent model has been developed. It accommodates 60 × 16 square grids of 6.67-km mesh size and uses a time step of 12 h. The governing equation in the model is the time-dependent partial differential equation with the five components of advection, diffusion, sources, settling, and regeneration. The physical advective transport is based on a set of computed daily averaged currents

which were generated by a CCIW hydrodynamic model (Simons 1976). A horizontal diffusion coefficient of $A = 2.5 \times 10^5$ cm^2/s was used for the diffusion term, in agreement with results from diffusion experiments for the length scale of 6.67 km and also confirmed by earlier chloride computations. Actual data were used for the source terms and for the settling process, a settling rate of 0.25 m/day was selected.

Under high wind conditions, the regeneration of phosphorus is attributed mainly to the wave action of water on the sediments. A formula is proposed which relates the regenerated amount of total phosphorus to wave oscillatory motion and sediment mean grain size. The wave characteristics were computed by SMB hindcasting techniques from a 3-h wind data. Known concepts and data in wave theory, sedimentology, and geochemistry have been incorporated in the formulation of this regeneration submodel.

This study is a first attempt at system modeling simulation, utilizing experimental results of the basic physical, chemical, and geological processes of transports and regeneration of phosphorus in a water body. It is shown that it is possible to reconstruct, by modeling techniques, much of the pattern of phosphorus distributions existing between cruises. In this sense, the physical transports and regeneration of phosphorus between cruises are quantitatively described by means of a mathematical model.

This model should not be considered as final. Despite the fact that it produces reasonably good simulation results for total phosphorus concentration under high wind conditions, it is still primitive and incomplete. Further efforts must be made to extend the model to simulate processes involving a system of various forms of phosphorus under stratified or unstratified conditions. When the complete model is verified, it will be a good management tool for assessing phosphorus disposal and removal practices.

The physical regeneration submodel is still crude because of a lack of both theoretical and experimental evidence on the relation between wave motions and sediment resuspension in actual lake environments. Improvement of the submodel will be possible only after in situ measurements and wave-tank experiments have been applied to clarify this relation.

The undetermined coefficients such as σ and K_r in the model were set in heuristic manner. Current efforts are directed towards establishing a mathematical and objective criterion for comparing the computed and observed results, so that judgements as to goodness of fit are not only reliant on visual comparison of two maps, but on a numerical objective criterion. This new study may give information regarding the optimal design of planning sampling stations over the lake.

Acknowledgments

We thank N. M. Burns and M. A. Donelan for helpful discussions and suggestions and J. P. Coakley who provided the wave hindcast computer program. T. M. Dick, M. Skafel, and B. G. Krishnappan critically read the manuscript and their assistance is gratefully acknowledged.

BURNS, N. M. 1976a. Nutrient budgets for Lake Erie, 1970. J. Fish. Res. Board Can. 33: 520–536.

1976b. Temperature, oxygen, and nutrient distribution patterns in Lake Erie, 1970. J. Fish. Res. Board Can. 33: 485–511.

BURNS, N. M., AND A. E. PASHLEY. 1974. In situ measurement of the settling velocity profile of particulate organic carbon in Lake Ontario. J. Fish. Res. Board Can. 31: 291–297.

COLE, A. L., AND J. C. AYERS. 1967. Wave hindcasts versus recorded waves. Final rep. U.S. Army Eng. Distr. Lake Surv. Coll. Eng., Dep. Meteorol. Oceanogr., Univ. Michigan, Ann Arbor, Mich. 88 p.

DAS, M. M. 1972. Mechanics of sediment suspension due to oscillatory water waves, p. 11.1–11.12. *In* H. W. Shen [ed.] Sedimentation. Univ. Colorado, Boulder, Colo.

GRAF, W. M. 1971. Hydraulics of sediment transport. McGraw-Hill, Inc., New York, N.Y. 513 p.

HOM-MA, M., AND K. HORIKAWA. 1962. Suspended sediment due to wave action. Proc. Conf. Coastal Eng., Mexico City. 8: 168–193.

IMBODEN, D. M. 1974. Phosphorus model of lake eutrophication. Limnol. Oceanogr. 19: 297–304.

INMAN, D. L. 1957). Wave-generated ripples in nearshore sands. U.S. Dep. Army, Corps Eng., Tech. Memo. 100: 42 p.

KEMP, A. L. W., R. L. THOMAS, C. I. DELL, AND J.-M. JAQUET. 1976. Cultural impact on the geochemistry of sediments in Lake Erie. J. Fish. Res. Board Can. 33: 440–462.

KING, C. A. M. 1972. Beaches and coasts. E. Arnold, London. 570 p.

KOMAR, P. D., AND M. C. MILLER. 1973. The threshold of sediment movement under oscillatory water waves. J. Sediment. Petrol. 43/4, 1101–1110.

1975. On the comparison between the threshold of sediment motion under waves and unidirectional currents with a discussion of the practical evaluation of the threshold. J. Sediment. Petrol. 45/1: 362–367.

KOMAR, P. D., R. H. NEUDECK, AND L. D. KULN. 1972. Observations and significance of deep-water oscillatory ripple marks on the Oregon continental shelf, p. 601–619. *In* D. P. Swift [ed.] Shelf sediment transport: process and pattern. Dowden, Inc., Stroudsburg, Pa.

LAM, D. C. L., AND T. J. SIMONS. 1976. Numerical computations of advective and diffusive transports of chloride in Lake Erie, 1970. J. Fish. Res. Board Can. 33: 537–549.

LEBON, G., AND J. LAMBREMONT. 1975. Détermination du flux d'érosion des sédiments cohésifs sous l'action de courants turbulents stationnaires. Presented at the International Sedimentological Congress, Nice (France). 9: 6 p.

LONSDALE, P., AND J. B. SOUTHARD. 1974. Experimental erosion of North Pacific clay. Mar. Geol. 17: M 51-M60.

MCCAVE, I. N. 1973. Transport escape of fine-grained sediment from shelf areas, p. 225–248. *In* D. P. Swift [ed.] Shelf sediment transport: process and pattern. Dowden, Inc., Stroundsburg, Pa.

MIGNIOT, C. 1968. Etude des propriétés physiques des différents sédiments très fins et de leur comportement sous des actions hydrodynamiques. Houille Blanche 7: 591–620.

MOORE, C. A., AND M. L. SILVER. 1973. Nutrient transport by sediment-water interaction. Water Resour. Cent. Res., Univ. Illinois, Urbana, Ill. Rep. 65: 51 p.

NIELSON, F. D. 1973. Behaviour of cohesive material from soil engineer's viewpoint, p. 14.1–14.28. *In* H. W. Shen [ed.] Environmental impacts on rivers (River mechanics III). Univ. Colorado, Boulder, Colo.

NRIAGU, J. O., AND C. I. DELL. 1974. Diagnetic formation of iron phosphates in recent lake sediments. Am. Mineral. 59: 934–946.

PARTHENIADES, E. 1971. Results of recent investigations on erosion and deposition of cohesive sediments, p. 20.1–20.39. *In* H. W. Shen [ed.] River mechanics II. Univ. Colorado, Boulder, Colo.

RIEDEL, P. H., AND J. W. KAMPHUIS. 1973. A shear plate for use in oscillatory flow. J. Hydraul. Res. 11: 137–156.

SIMONS, T. J. 1976. Continuous dynamical computations of water transports in Lake Erie for 1970. J. Fish. Res. Board Can. 33: 371–384.

SOUTHARD, J. B., R. A. YOUNG, AND C. D. HOLLISTER. 1971. Experimental erosion of calcareous ooze. J. Geophys. Res. 76/24: 5903–5909.

STERNBERG, R. W. 1972. Prediction of initial motion and bedload transport of sediment particles in the shallow marine environment, p. 61–82. *In* D. P. Swift [ed.] Shelf sediment transport: process and pattern. Dowden, Inc., Stroudsburg, Pa.

THOMAS, R. L., J.-M. JAQUET, A. L. W. KEMP, AND C. F. M. LEWIS. 1976. Surficial sediments of Lake Erie. J. Fish. Res. Board Can. 33: 385–403.

U.S. COASTAL ENGINEERING RESEARCH CENTER. 1973. Shore protection manual. Fort Belvoir, Va.

WILLIAMS, J. D. H., J.-M. JAQUET, AND R. L. THOMAS. 1976. Forms of phosphorus in the surficial sediments of Lake Erie. J. Fish. Res. Board Can. 33: 413–429.

WILLIAMS, J. D. H., AND T. MAYER. 1972. Effects of sediment diagenesis and regeneration of phosphorus with special reference to Lakes Erie and Ontario, p. 281–315. *In* H. F. Allen and J. R. Kramer [ed.] Nutrients in natural waters. John Wiley and Sons, Inc., New York, N.Y.

YALIN, M. S., and R. C. RUSSEL. 1966. Shear stresses due to long waves. J. Hydraul. Res. 412: 55–98.

A Phosphorus Budget for Lake Erie

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Burns, N. M., J. D. H. Williams, J.-M. Jaquet, A. L. W. Kemp, and D. C. L. Lam. 1976. A phosphorus budget for Lake Erie. J. Fish. Res. Board Can. 33: 564–573.

A budget for phosphorus retention in Lake Erie during 1970 has been drawn up. Phosphorus associated with sand-sized particles (>63 μm) was excluded. Retention of apatite phosphorus (AP) and nonapatite phosphorus (NAP) within each of the three basins of the lake (Western, Central, and Eastern) was estimated in each case by two independent methods. The two estimates of retention of total phosphorus within the entire lake agreed to within 9.5% of their mean. The estimates of AP retention within the Western basin also agreed well. However, the input–output balance estimates of AP retention in the Central basin and of NAP retention in the Western and Central basins greatly exceeded the corresponding sedimentation values, while in the Eastern basin this pattern was reversed. The AP discrepancy was attributed to unobserved transport of AP from the Central to the Eastern basin via nearshore currents past Long Point, while the NAP discrepancies were largely attributed to eastward movement of NAP through the lake within about 2 m of the lake bottom. The total amount of phosphorus retained within the lake in 1970 was 41,000 metric tons, or about 92% of input. AP of natural origin and inert nature accounted for approximately 44% of the total phosphorus retained. A reduction in anthropogenic phosphorus input of about 15,000 metric tons/yr, as called for in the Great Lakes Water Quality Agreement, will reduce present annual NAP loading to close to pre-1850 values.

Burns, N. M., J. D. H. Williams, J. -M. Jaquet, A. L. W. Kemp, and D. C. L. Lam. 1976. A phosphorus budget for Lake Erie. J. Fish. Res. Board Can. 33: 564–573.

Les auteurs ont établi un budget pour la rétention du phosphore dans le lac Erié durant 1970. Le phosphore associé aux particules de la grosseur de grains de sable (>63 μm) a été exclu. La rétention du phosphore associé à l'apatite (PA) et du phosphore non-apatite (PNA) dans chacun des trois bassins du lac (occidental, central et oriental) a été estimée dans chaque cas selon deux méthodes indépendantes. Les deux estimés de rétention du phosphore total dans le lac entier sont en accord dans une limite de 9.5% de leur moyenne. Les estimés de rétention du PA dans le bassin oriental s'accordent également bien. Cependant, les estimés de bilan entrée–sortie de la rétention de PA dans le bassin central et de rétention de PNA dans les bassins occidental et central dépassent de beaucoup les valeurs de sédimentation correspondantes, alors que dans le bassin oriental la situation est inverse. On attribue la différence de PA à un transport non observé de PA du bassin central au bassin oriental par les courants littoraux passant le long de Pointe Longue, alors que les différences de PNA sont en grande partie attribuables au mouvement vers l'est de PNA à travers le lac à moins d'environ 2 m du fond du lac. La quantité totale de phosphore retenu dans le lac en 1970 est de 41,000 tonnes métriques, soit environ 92% de l'apport. Le PA d'origine naturelle et de nature inerte représente environ 44% du phosphore total retenu. Une réduction de l'apport de phosphore d'origine humaine d'environ 15,000 tonnes métriques/a. selon les termes de l'Accord sur la qualité de l'eau des Grands lacs, diminuera la présente charge annuelle de PNA à des niveaux se rapprochant de ceux d'avant 1850.

Accepted November 18, 1975

Accepté le 18 novembre 1975

The use of data on phosphorus in Lake Erie, included in four other studies in this issue, has made it possible to draw up two independent estimates of phosphorus retention in the lake. These two retention estimates have been compared with each other and an attempt has been made to draw up a complete phosphorus budget for the lake. This whole-lake budget has also been subdivided into basin budgets. Data on water and air-borne phosphorus inputs to and outputs from the lake (Burns 1976a) have been combined with estimates of erosional loadings of fine-grained materials (Kemp et al. 1976) and measurements of the phosphorus content of shoreline materials (Williams et al. 1976a) to obtain an estimate of the quantity of phosphorus retained in the lake. A second estimate of phos-

Printed in Canada (J4085)
Imprimé au Canada (J4085)

phorus retained in the lake was obtained from the sedimentation rates of the fine-grained materials in the depositional basins of the lake (Kemp et al. 1976).

From an examination of the forms of phosphorus in both the shoreline materials and sediments of the lake, Williams et al. (1976a, 1976b) were able to partition the phosphorus loadings into estimates of apatite and nonapatite inputs to the lake waters and sediments. This information has been utilized in estimating both apatite and nonapatite budgets for the lake. The division of the total phosphorus budget into budgets for the two different phosphorus forms is necessary because it appears that apatite does not enter into the phosphorus cycle of Lake Erie (Burns 1976a; Williams et al. 1976a).

Methods of Budget Calculation

An explanation of the various symbols used in the different budgets is given in Table 1. The data, assumptions, and methods used in the calculation of the two independent whole-lake budgets are shown in Table 2. The first estimate of the phosphorus retained in the sediments is the result of subtracting the phosphorus outputs from the total phosphorus inputs to the lake and is referred to as the input–output balance (IOB) estimate of retention. The second estimate of this quantity, derived from an inspection of sediment cores, was calculated by Kemp et al. (1976) and is referred to as the sedimentation estimate. The details of the calculation of the apatite and nonapatite budgets for each of the three basins are shown in Tables 3–8. Nonapatite phosphorus (NAP) is defined as that fraction of the total phosphorus (TP), in either a sample or budget, which is not apatite phosphorus (AP).

Only the soluble and fine-grained forms of phosphorus have been considered in this study. "Fine-grained" refers here to materials whose equivalent spherical diameter is less than 62.5 μm (4 ϕ). Coarser inorganic materials are excluded, partly because there are as yet no reliable estimates of the amounts of these materials entering the lake but largely because it is almost certain that the coarse-grained inorganic forms of phosphorus, containing only apatite, are not subject to dissolution in the waters of the lake (Williams et al. 1976a; Burns 1976a).

The values for the average AP and NAP contents of the Western basin (W.B.) sediments are the arithmetic mean of the A-P and NA-P values (Table 1) of the eight surficial sediments from this basin which were finer than 4 ϕ (figure 1 of Williams et al. 1976b). Similar values were calculated for the Central basin (C.B.) and Eastern basin (E.B.) based on 17 and 9 samples, respectively.

Data inputs and derived values used in constructing Tables 2–8 are denoted by the symbols D1, D2, etc. in the order of introduction or derivation. In addition, certain assumptions were made and these are denoted by the symbols A1, A2, etc., again in the order of introduction. The four major data sources are denoted by the symbols R1–R4 (Table 1).

TABLE 1. Explanation of symbols.

Symbol	Explanation
A	Assumption
AP	Apatite phosphorus, concept or physical reality
A-P	Apatite phosphorus, analytical estimate of
C.B.	Central basin
D	Datum
E.B.	Eastern basin
IOB	Input–output balance
NAP	Nonapatite phosphorus, concept or physical reality
NA-P	Nonapatite phosphorus, analytical estimate of
ppm	Mg of X per kg of oven-dried sediment (parts per million)
R	Reference
R1	Burns 1976a
R2	Kemp et al. 1976
R3	Williams et al. 1976b
R4	Williams et al. 1976a
TP	Total phosphorus, concept or physical reality
T-P	Total phosphorus, analytical estimate of
W.B.	Western basin

Results

The net annual retention of total phosphorus in Lake Erie was around 41,000 tons in 1970 (Table 2). About two thirds of this total entered the lake via inflowing rivers and municipal outlets or from the atmosphere, and the remainder as particulate phosphorus derived from eroding shoreline bluffs. The difference between the two estimates of retention of total phosphorus, as measured by the input–output balance and by sedimentation, differed by 9.5% of their mean value. This agreement was considered satisfactory in view of the difficulties in estimating many of the quantities involved in this study. Table 9 summarizes the retention of AP and NAP in each basin and in the whole lake. If the average values of retention are used, AP accounts for 44% of the total phosphorus retained in the lake at the present time. According to Kemp et al. (1976), the rate of sedimentation of total phosphorus prior to 1850 was 24,500 metric tons/yr, compared to 39,100 metric tons/yr for 1970. If the rate of sedimentation of apatite is assumed to be the same prior to 1850 as at present (Kemp et al. 1976; Williams et al. 1976a) AP accounted for 73% of the total phosphorus retained in the lake

TABLE 2. Total phosphorus budget for Lake Erie in the early 1970s.

Assumption or data item		Data	Subtotal (*metric tons/yr*)	Major Total (*metric tons/yr*)	Reference
A1	All shoreline erosional inputs of fine-grained material to the lake have AP and NAP concentrations which are the same as those of the C.B. northshore bluffs				
D1	T-P value of northshore bluffs (mean of 14 samples)	576 ppm			R4
D2	Sediment input due to shoreline erosion entire lake	25.7×10^6			R2
D3	TP input to entire lake due to shoreline erosion [= (D1) × (D2)]		14,800		
A2	Dredged spoils are not a separate sediment input, being a redistribution of material already within the lake				
D4	TP input due to dredging		0		A2
D5	Gross input of water-borne and air-borne phosphorus into entire lake		32,000		R1
D6	Export of phosphorus out of lake		3,800		R1
D7	IOB of water-borne and air-borne phosphorus into entire lake [= (D5) − (D6)]		28,200		
D8	Input of TP into entire lake, all sources [= (D3) + (D4) + (D5)]		46,800		
D9	IOB of TP for entire lake, all sources [= (D3) + (D4) + (D7)]			43,000	
D10	TP sedimentation, entire lake			39,000	R2
			Avg retention	41,000	
D11	Discrepancy between estimates of retention of TP for entire lake, based on net IOB compared with sedimentation [= (D9) − (D10)]			+3,900	
D12	First estimate of percentage retention of TP in the entire lake, based on IOB [= (D9)/(D8) × 100]		91.9		
D13	Second estimate of percentage retention of TP in the entire lake, based on sedimentation (= (D10)/((D10) + (D6)) × 100)		91.1		

prior to 1850. This value is in agreement with the estimate of Williams et al. (1976a), obtained by independent methods, that AP accounted for 60–80% of the total phosphorus sedimented in the lake during this time period.

The most striking feature of the results shown in Table 9 is that the differences between each of the estimates of retention for AP and NAP are generally larger for the individual basins than for the whole lake. Only in the case of AP retention in the W.B. is there fair agreement between the IOB and sedimentation estimates. The estimate of AP input to the C.B. is much greater than the quantity of apatite found in the sediments of that basin; the reverse situation is seen to exist in the E.B. (Table 9).

In the case of NAP, it can be seen that a large difference between the IOB sediment loading estimate and the sedimentation estimate is found in the W.B. A similar but small difference is seen to exist in the C.B. estimates. However, in the E.B., the estimate of NAP sedimentation is 7800

TABLE 3. Apatite phosphorus (AP) budget for the W.B.

Assumption of data item		Data	Total (*metric tons/yr*)	Reference
A3	The entire AP input of any basin is retained within that basin			see text
A4	The AP content of all fine-grained sediments entering the lake is the same as the northshore bluff content			see text
D14	Total fine-grained sediment input, W.B.	3.7×10^6 metric tons/yr		R2
D15	A-P for northshore bluffs (mean of 14 samples)	545 ppm		A4, R4
D16	IOB for AP, W.B. [= (D14) × (D15)]		2000	A3
D17	T-P for fine-grained W.B. sediments (mean of eight samples)	1157 ppm		R3
D18	A-P, ditto	354 ppm		R3
D19	A-P as a proportion of T-P, W.B. [= (D18)/(D17)]	0.306		
D20	TP sedimentation, W.B.	5800 metric tons/yr		R2
D21	AP sedimentation, W.B. [= (D19) × (D20)]		1800	
D22	Discrepancy between estimates of retention of AP for W.B., based on IOB compared with sedimentation [= (D16) − (D21)]		+200	

TABLE 4. Apatite phosphorus (AP) budget for the C.B.

Assumption or data item		Data	Total (*metric tons/yr*)	Reference
D23	Total fine-grained sediment input, C.B.	24.7×10^6 metric tons/yr		R2
D24	IOB for AP, C.B. [= (D15) × (D23)]		13,500	A3
D25	T-P for fine-grained C.B. sediments (mean of 17 samples)	881 ppm		R3
D26	A-P ditto	431 ppm		R3
D27	A-P as a proportion of T-P, C.B. [= (D26)/(D25)]	0.489		
D28	TP sedimentation, C.B.	11,700 metric tons/yr		R2
D29	AP sedimentation, C.B. [= (D27) × (D28)]		5,700	
D30	Discrepancy between estimates of retention of AP for C.B., based on IOB compared with sedimentation [= (D24) − (D29)]		+7,800	

metric tons/yr greater than the IOB loading estimate to that basin.

Discussion

Discrepancies Between Retention Estimates

The assumption that the entire AP input to a basin is held within that basin (assumption A3) is apparently correct for the W.B., but is clearly not true for the C.B. This is not surprising since Kemp et al. (1976) have shown that the source of most of the sediment accumulating in the E.B. is from shoreline erosion on the north shore of the C.B. Clearly, much of the AP that enters Lake Erie from this source is carried to the E.B. by the same mechanism that transported the other

TABLE 5. Apatite phosphorus (AP) budget for the E.B.

Assumption or data item		Data	Total (*metric tons/yr*)	Reference
D31	Total sediment input, E.B.	1.4×10^6 metric tons/yr		R2
D32	IOB for AP, E.B. [= (D15) × (D31)]		800	A3
D33	T-P for fine-grained E.B. sediments (mean of nine samples)	946 ppm		R3
D34	A-P, ditto	530 ppm		R3
D35	A-P as a proportion of T-P, E.B. [= (D34)/(D33)]	0.560		
D36	TP sedimentation, E.B.	21,600 metric tons/yr		R2
D37	AP sedimentation, E.B. [=(D35) × (D36)]		12,100	
D38	Discrepancy between estimates of retention of AP for E.B., based on IOB compared with sedimentation [= (D32) − (D37)]		−11,300	

TABLE 6. Nonapatite phosphorus (NAP) budget for the W.B.

Assumption or data item		Data	Subtotal (*metric tons/yr*)	Major total (*metric tons/yr*)	Reference
D39	Gross input of water and air-borne TP into W.B.		23,700		R1
D40	Sediment input due to shoreline erosion, W.B.	0.4×10^6 metric tons/yr			R2
D41	TP input to W.B. due to shoreline erosion [= (D1) × (D40)]		200		A1
D42	NAP input, W.B. [= (D39) + (D41) − (D16)]		21,900		
D43	NAP exported from W. to C.B.		6,200		R1, A3
D44	NAP retention by IOB, W.B. [= (D42) − (D43)]			15,700	
D45	NAP sedimentation, W.B. [=(D20) − (D21)]			4,000	
D46	Discrepancy between estimates of retention of NAP for W.B., based on IOB compared with sedimentation [= (D44 − (D45)]			+11,700	

eroded bluff materials. The northeast shoreline area of the C.B. is a high-energy zone (Thomas et al. 1976) and during periods of heavy southwest winds, fine-grained material is resuspended and moved eastward by strong nearshore easterly currents (Simons 1976).

It is known that the currents in the northeastern C.B. often run parallel with and fairly close to the shoreline, with a substantial portion of the volume of water flow into the E.B. occurring through this zone. The northern water sampling station on the C.B.–E.B. boundary was not in this high-transport zone, being situated about 13 km from the shoreline (see figure 1 in Burns 1976a); thus little of the AP being transported to the E.B. was observed in the water sampled during the ship surveys. This resulted in an underestimate by Burns (1976a) of the AP transport from the C.B. to the E.B.

It is not possible to explain the difference between the IOB estimate of NAP retention in the W.B. and the NAP sedimentation estimate by

TABLE 7. Nonapatite phosphorus (NAP) budget for the C.B.

Assumption or data item		Data	Subtotal (*metric tons/yr*)	Major total (*metric tons/yr*)	Reference
D47	Input of water-borne and air-borne TP into C.B.		5,800		R1
D48	Sediment input to C.B. due to shoreline erosion	24.3×10^6 metric tons/yr			R2
D49	TP input to C.B. due to shoreline erosion [= (D1) × (D48)]		14,000		A1
D50	NAP input, C.B. [= (D43 + (D47) + (D49) − (D24)]		12,500		
D51	NAP exported from C. to E.B.		3,200		R1, A3
D52	NAP retention by IOB, C.B. [= (D50) − (D51)]			9,300	
D53	NAP sedimentation, C.B. [= (D28) − (D29)]			6,000	
D54	Discrepancy between estimates of retention of NAP for C.B., based on IOB compared with sedimentation [= (D52) − (D53)]			+3,300	

TABLE 8. Nonapatite phosphorus (NAP) budget for the E.B.

Assumption or data item		Data	Subtotal (*metric tons/yr*)	Major total (*metric tons/yr*)	Reference
D55	Input of water and air-borne TP into E.B.		2500		R1
D56	Sediment input due to shoreline erosion, E.B.	1.0×10^6 metric tons/yr			R2
D57	TP input to E.B. due to shoreline erosion [= (D1) × (D56)]		600		A1
D58	NAP input, E.B. [= (D51) + (D55) + (D57) − (D32)]		5500		
D59	NAP exported from E.B. via Niagara and Welland		3800		A3, R1
D60	NAP retention by IOB, E.B. [= (D58) − (D59)]			1700	
D61	NAP sedimentation, E.B. [= (D36) − (D37)]			9500	
D62	Discrepancy between estimates of retention of NAP for E.B., based on IOB compared with sedimentation [= (D60) − (D61)]			−7800	

using arguments similar to those outlined above. The water sampling stations on the boundary between the Western and Central basins were situated in the area where the volume of water transport between the two basins is great. Further, in contrast to the C.B., more than 90% of the fine-grained sediment loaded into the W.B. is found to remain in that basin (Kemp et al. 1976; Burns 1976b).

The quantities of NAP which have apparently disappeared from the W.B. system and appeared in the E.B. are much greater than the whole-lake phosphorus budget imbalance and cannot be considered to be the result of errors in the method

TABLE 9. Summary of all retention estimates.[a]

		Basin			Whole lake	Average of two retention estimates for whole lake
		Western	Central	Eastern		
AP	IOB	2,000	13,500	800	16,300	
	Sedimentation	1,800	5,700	12,100	19,600	18,000
	Discrepancy	+200	+7,800	−11,300	−3,300	
NAP	IOB	15,700	9,300	1,700	26,700	
	Sedimentation	4,000	6,000	9,500	19,500	23,100
	Discrepancy	+11,700	+3,300	−7,800	+7,200	
TP	IOB	17,700	22,800	2,500	43,000	
	Sedimentation	5,800	11,700	21,600	39,100	41,100
	Discrepancy	+11,900	+11,100	−19,100	+3,900	

[a]Equation used in computing quantities:
(net input to basin) − (input to sediments) = discrepancy
negative discrepancy (−) = excess deposition in sediments
positive discrepancy (+) = deficient deposition in sediments

of budget calculation. Further evidence exists to indicate that the disappearance of phosphorus from the W.B. waters is real. Figure 1 shows a contour map of the average total phosphorus concentrations which were observed during 10 surveys of Lake Erie (Burns 1976a). The net water transport is from west to east and the decrease in total phosphorus concentration in this direction in the W.B., from 1.8 to 0.8 μmol P liter^{-1}, reflects the disappearance of phosphorus from the water. A smaller loss of phosphorus from the water of the C.B. is obvious from the decrease in concentration of total phosphorus from 0.8 to 0.5 μmol liter^{-1} in the easterly direction along the lake. There is no decrease in the phosphorus concentration in the E.B. waters in this direction.

The phosphorus concentration values shown in Fig. 1 refer to the water column from the water surface to 2.0 m above the bottom. The zone from 2.0 m above the sediment to the sediment–water interface was not sampled from the ship. Thus it would seem that there were two zones of the lake which were inadequately sampled during the ship surveys of the lake; these being the nearshore zone and the zone from the sediment to 2.0 m above the sediment. It is probable that there was a large unobserved transport of phosphorus from west to east in the lake through both of these zones.

As discussed above, it is likely that much of the sediment and AP transported from the C.B. to the E.B. moved along the nearshore zone. However, the zone of major water transport through the W.B.–C.B. boundary is not in the nearshore zone but in the deeper offshore zones (Lam and Simons 1976). Water sampling stations were situated in these zones (Burns 1976a) and the phosphorus fluxes calculated on the basis of the observed concentrations are confirmed by Lam and Jaquet (1976). It is more likely that the majority of the unobserved phosphorus transport from the W.B. to the C.B. occurred through the other unmonitored zone of the lake, namely between the sediment–water interface and 2.0 m from the sediments.

Although the variation in suspended solids concentration is seen to be small in the zone from the surface to 2.0 m above the bottom (Burns 1976b), it is probable that strong concentration gradients did occur within the bottom water layer as has been observed in the nearshore marine environment by Hom-ma and Horikawa (1962). It is not unreasonable, therefore, to assume the episodic occurrence in Lake Erie of a bottom-water layer, enriched in suspended solids and similar to the so-called "nepheloid layer" repeatedly observed in oceans (Gibbs 1974). Significant lateral transport of material, including particulate and colloidal phosphorus, could occur within this layer but would not be included in computations of flux between basins because the water within this layer was never sampled and the high concentrations of various suspended materials were not observed.

Either iron-coated clay minerals or particulate organic carbon could act as a carrier for NAP transport. Phosphorus could be adsorbed onto fine sedimentary clay particles, which move downstream through the lake by the process of frequent resuspension and settling. However, this cannot account for the export of NAP out of the W.B. because very little of the sediment of that basin is exported. Neither is NAP transport by clay

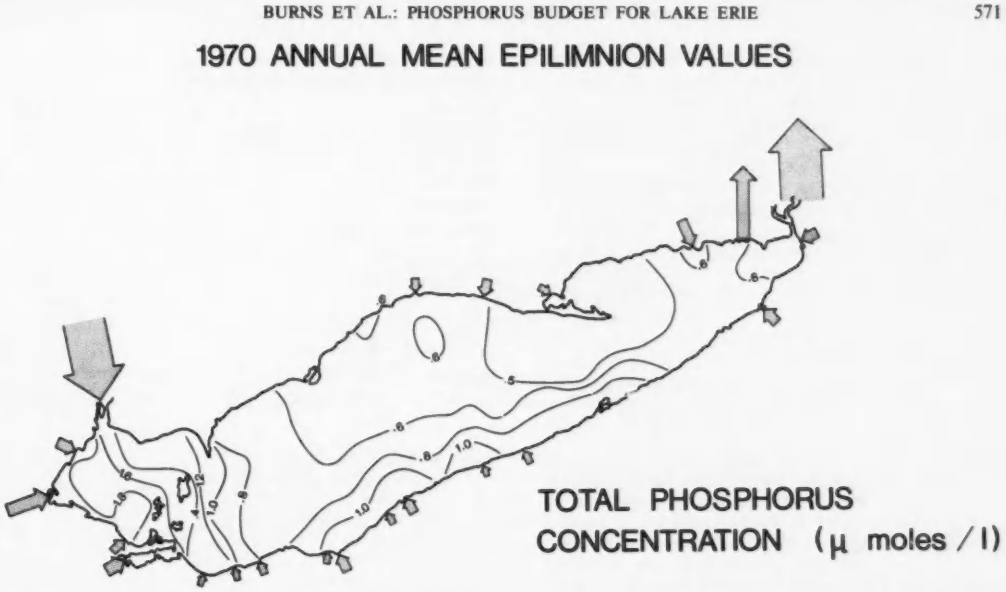

FIG. 1. Distribution pattern of the average phosphorus concentrations observed during 10 surveys of Lake Erie from April to December 1970.

particles a totally adequate explanation for the phosphorus transfer between C.B. and E.B. because the sediment and phosphorus sources are remote from each other.

An alternative explanation for the transport of NAP is provided via the observation of divers that a thin layer of flocculent organic material often occurs at the sediment–water interface in Lake Erie (Burns and Ross 1972 and Burns personal communication). It is possible that this material is formed from organic matter decomposing at the interface. The material is assumed to have a density greater than that of fresh organic matter, but not great enough to be easily incorporated into the sediment. This material could be similar to the Hg-rich organic material described by Cline et al. (1973) in Detroit river. This floc would contain phosphorus, possibly in colloidal form (Lean 1973), but little clay. Owing to its comparatively low density, the floc would move along the bottom of the lake more easily than the denser inorganic particles, and it is assumed that this mechanism accounts for much of the transport of NAP through the lake. The high sedimentation rate of organic carbon in the E.B. (Kemp et al. 1976) tends to confirm that a net eastward transport of organic floc does occur in Lake Erie.

According to Burns et al. (1976), phosphorus fluxes through basin boundaries varied throughout the year with larger amounts being transported in November and December 1970. These results are confirmed by Lam and Jaquet (1976, table 5) who computed, with their model, advective and diffusive fluxes of phosphorus for the survey intervals 8–9 and 9–10 which were periods characterized by stormy conditions. This demonstrates that in addition to "normal" transport via organic floc, phosphorus fluxes between basins can be substantially increased following physical regeneration by storm-induced resuspension. However, much of the phosphorus transport between basins during storms as calculated by Burns et al. (1976) included the resuspended phosphorus because, under these conditions, resuspension was not restricted to the nearshore and near-sediment zones but occurred throughout the water column and over the entire lake (Lam and Jaquet 1976). Thus the water containing these high concentrations was sampled during the ship surveys and the results were included in the phosphorus flux calculations.

IMPLICATIONS OF LOADING REDUCTIONS

Using the estimate that AP constituted 73% of the precolonial loading of phosphorus to the lake but made up only 44% of the 1970 loadings, it is calculated that the NAP loading to the lake sediments was in the order of 6600 metric

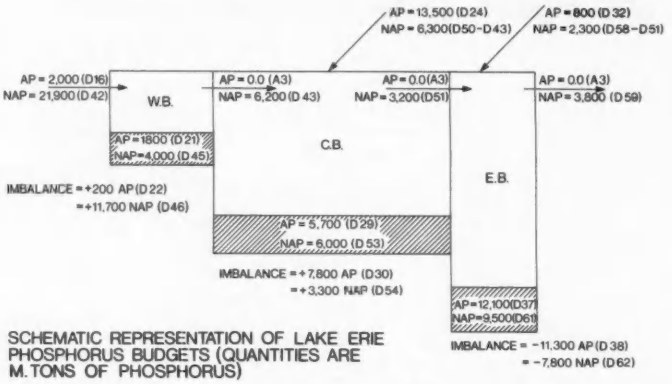
AP = 13,500 (D 24)
NAP = 6,300 (D 50 - D 43)
AP = 800 (D 32)
NAP = 2,300 (D 58 - D 51)
AP = 2,000 (D 16)
NAP = 21,900 (D 42)
W.B.
AP = 0.0 (A 3)
NAP = 6,200 (D 43)
AP = 0.0 (A 3)
NAP = 3,200 (D 51)
AP = 0.0 (A 3)
NAP = 3,800 (D 59)
AP = 1800 (D 21)
NAP = 4,000 (D 45)
C.B.
E.B.
IMBALANCE = +200 AP (D 22)
= +11,700 NAP (D 46)
AP = 5,700 (D 29)
NAP = 6,000 (D 53)
IMBALANCE = +7,800 AP (D 30)
= +3,300 NAP (D 54)
AP = 12,100 (D 37)
NAP = 9,500 (D 61)
IMBALANCE = −11,300 AP (D 38)
= −7,800 NAP (D 62)
SCHEMATIC REPRESENTATION OF LAKE ERIE PHOSPHORUS BUDGETS (QUANTITIES ARE M. TONS OF PHOSPHORUS)

FIG. 2. A schematic summary of the data inputs and derived water–sediment phosphorus budgets. The numbers in parentheses refer to assumptions and data items described in Tables 2–8; a positive (+) imbalance refers to the situation where more phosphorus was lost from the overlying waters than was observed in the sediments; a negative (−) imbalance situation results from less phosphorus being lost from the water than was found in the underlying sediments.

tons/yr prior to 1850, with the 1970 loading having been approximately 23,100 metric tons/yr. This represents an increase of 16,500 metric tons/yr in the NAP loading rate from precolonial times to the present. The Great Lakes Water Quality Agreement (U.S. Department of State 1972) calls for a reduction in the water-borne phosphorus loading from 29,300 metric tons/yr in 1970 to 14,600 metric tons/yr in 1976 which level is then to be maintained indefinitely into the future. This would imply a reduction of 14,700 metric tons/yr in the anthropogenic loading of phosphorus to the lake. Since the phosphorus derived from man's activities is nearly all in the form of NAP, the full implementation of the Great Lakes Water Quality Agreement would decrease the loading of NAP to the lakes by approximately 14,700 metric tons/yr to a level of 8400 metric tons/yr. This loading is close to the precolonial loading of NAP (6600 metric tons/yr) and if the loading reductions were implemented, a significant improvement in the trophic state of the lake could be expected once the appropriate sediment–water phosphorus equilibria had been attained.

Summary and Conclusions

The values and data references that are shown in Fig. 2 provide an adequate summary of the results of this investigation.

The large apatite (AP) deficiency in the C.B. sediments and the AP surplus in the E.B. sediments can be explained by proposing that the majority of the material transported from the C.B. to the E.B. was first resuspended by wave-induced turbulence along the northeast shoreline area of the C.B. as a result of southwest winds and then subsequently moved eastward by the strong nearshore currents resulting from these winds. The material undergoing transport by this region of the C.B.–E.B. boundary were not sampled during the ship surveys.

A comparison of the (IOB) and the sedimentation estimate for NAP in the W.B. showed a massive loss from the W.B. waters without the corresponding amount of phosphorus being found in the W.B. sediments. However, a large part of the NAP lost from the W.B. system was found in the E.B. sediments. It is postulated that large quantities of NAP were transported from the W.B. through the C.B. and into the E.B. in the form of a resuspended organic floc moving in the downstream direction usually within 2.0 m of the sediment–water interface. It is further proposed that this organic material has undergone some humification resulting in the material having a density significantly greater than that of fresh organic matter. This would cause the organic floc material to settle quickly and remain near the sediment–water interface. These assumptions lead to the conclusion that the transport of large quantities of NAP to the E.B. would result in the simultaneous transport of large quantities of organic material to the E.B. sediments. Evidence to support these assumptions is found in

the fact that while the waters of the E.B. have the lowest production of organic material in the lake, the sediments of this basin were found to have the highest loading of organic carbon.

It is estimated that the precolonial loading of nonapatite phosphorus (NAP) to Lake Erie was in the order of 6600 metric tons per annum and that the full implementation of the Great Lakes Water Quality Agreement would reduce the present loading of water-borne NAP from approximately 21,900 to 8200 metric tons/yr. Since this lower value is close to the precolonial loading of NAP, the lake could be expected to show considerable improvement if the Water Quality Agreement is ever fully implemented.

BURNS, N. M. 1976a. Nutrient budgets for Lake Erie, 1970. J. Fish. Res. Board Can. 33: 520–536.

1976b. Temperature, oxygen, and nutrient distribution patterns in Lake Erie, 1970. J. Fish. Res. Board Can. 33: 485–511.

BURNS, N. M., AND C. ROSS. 1972. Oxygen–Nutrient relationships within the Central Basin of Lake Erie, p. 85–119. *In* N. M. Burns and C. Ross [ed.] Project Hypo. Can. Cent. Inland Waters Pap. No. 6.

BURNS, N. M., F. ROSA, AND C. H. CHAN. 1976. Lake Erie water chemistry data 1970–1971. Can. Cent. Inland Waters Pap. 16.

CLINE, J. T., J. L. HILLSON, AND C. B. UPCHURCH. 1973. Mercury mobilization as an organic complex. Proc. Conf. Int. Assoc. Great Lakes Res. (IAGLER) 16: 233–242.

GIBBS, R. J. [ed.] 1974. Suspended solids in water. Plenum Press, New York, N.Y. 320 p.

HOM-MA, M., AND K. HORIKAWA. 1962. Suspended sediment due to wave action. Proc. Conf. Coastal Eng. Mexico City 8: 168–192.

KEMP, A. L. W., R. L. THOMAS, C. I. DELL, AND J.-M. JAQUET 1976. Cultural impact on the geochemistry of sediments in Lake Erie. J. Fish. Res. Board Can. 33: 440–462.

LAM, D. C. L., AND J.-M. JAQUET. 1976. Computations of physical transport and regeneration of phosphorus in Lake Erie, fall 1970. J. Fish. Res. Board Can. 33: 550–563.

LAM, D. C. L., AND T. J. SIMONS. 1976. Numerical computations of advective and diffusive transports of chloride in Lake Erie, 1970. J. Fish. Res. Board Can. 33: 537–549.

LEAN, D. R. S. 1973. Phosphorus dynamics in lake water. Science 179: 678–680.

SIMONS, T. J. 1976. Continuous dynamical computations of water transports in Lake Erie for 1970. J. Fish. Res. Board Can. 33: 371–384.

THOMAS, R. L., J.-M. JAQUET, A. L. W. KEMP, AND C. F. M. LEWIS. 1976. Surficial sediments of Lake Erie. J. Fish. Res. Board Can. 33: 385–403.

U.S. DEPARTMENT OF STATE. 1972. Great Lakes Water Quality Agreement with annexes and texts and terms of reference between the U.S.A. and Canada. U.S. Gov. Print. Off., Washington, D.C. T1AS7312.

WILLIAMS, J. D. H., T. P. MURPHY, AND T. MAYER. 1976a. Rates of accumulation of phosphorus forms in Lake Erie sediments. J. Fish. Res. Board Can. 33: 430–439.

WILLIAMS, J. D. H., J.-M. JAQUET, AND R. L. THOMAS. 1976b. Forms of phosphorus in the surficial sediments of Lake Erie. J. Fish. Res. Board Can. 33: 413–429.

Bacterial Distributions in Lake Erie (1967, 1970)

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Rao, S. S., and B. K. Burnison. 1976. Bacterial distributions in Lake Erie (1967, 1970). J. Fish. Res. Board Can. 33: 574–580.

Distribution of heterotrophic and coliform bacteria in Lake Erie is a first step in determining the importance of bacteria in the lake. The abundance of these organisms was greatest in the Western basin. Offshore bacterial densities from the Central and Eastern basins were lower than the inshore areas. Detroit River, Grand River (Ontario), and the areas close to Erie, Pennsylvania and Cleveland, Ohio showed the highest bacterial densities throughout the study periods.

Rao, S. S., and B. K. Burnison. 1976. Bacterial distributions in Lake Erie (1967, 1970). J. Fish. Res. Board Can. 33: 574–580.

La distribution des bactéries hétérotrophes et coliformes dans le lac Erié est un premier pas lorsqu'il s'agit de déterminer l'importance des bactéries dans le lac. Ces organismes se trouvent en plus grande abondance dans le bassin occidental. Les densités bactériennes du large dans les bassins central et oriental sont plus faibles que celles des régions littorales. La rivière Détroit, la Grande rivière (Ontario) et les régions du voisinage d'Erié, Pennsylvanie, et de Cleveland, Ohio, montrent les plus hautes densités bactériennes durant toutes les périodes d'étude.

Accepted November 18, 1975 Accepté le 18 novembre 1975

The significance of the role of bacteria in lakes has been difficult to ascertain. There are many different physiological types of bacteria in natural waters, ranging from strict chemoautotrophs to phototrophs and to heterotrophs. The bacteria can be of allochthonous or autochthonous origin.

The present study is to be viewed only as the first step in understanding the role of heterotrophic bacteria in the Great Lakes. There are two standard methods for enumerating bacterial populations: 1) Direct counts can be made by filtering a known volume of water through a membrane filter. The organisms retained are stained and counted microscopically. These counts include both viable and dead cells. Sometimes a "value judgement" has to be made by the observer whether the object is a bacterial cell or a small detrital particle; 2) Plate counts are made by placing appropriate dilutions of the sample onto an agar-based nutrient medium. Only those bacteria capable of growing on this medium will be subsequently counted after a few days incubation. Recently, effort has again been made to explain the differences between plate counts and direct counts made by electronic particle counters (Ishida and Kadota 1975). Use of selective media allows certain types of bacteria to be counted quantitatively as, for example, coliforms. However, it is widely known that plate counts are usually not quantitative but they do give some insight into the distribution pattern of selected microbes. It must be noted that those bacteria that grow on the nutrient-rich agar plates may comprise from 50% (S. S. Rao unpublished data) to less than 1% of the total bacterial count obtained by direct count or even less if compared with scanning electron microscope counts (Overbeck 1974). Even under ideal conditions, the total number of heterotrophic bacteria will be less than the total bacterial population. However, the proportion of heterotrophs found near municipal and industrial sewage outfalls is higher than that found in offshore areas. This is probably a reflection of the higher "dissolved" organic carbon content of the nearshore water. These organic compounds may be the result of 1) organics in the sewage effluent; or 2) the organics excreted by phytoplankton which have developed because of the inflow of inorganic nutrients. Thus these organisms do provide an indication of the area of the lake affected by rivers and municipal discharges.

Schmidt (1973) has stated, "the common as-

Printed in Canada (J4070)
Imprimé au Canada (J4070)

sumption that numbers reflect activity is noted as probably so under some conditions, but as dangerously fallacious in perhaps most conditions." Recent methodology advances in microbial ecology have reemphasized that it is the in situ rate at which the bacteria perform their metabolic activities that is important in addition to their numbers.

In the past decade, the in situ activities of the heterotrophic bacteria have been under intensive study. The various methods that have been used are the measurement of O_2-consumption, CO_2-production, heterotrophic CO_2-fixation, and the uptake kinetics of labeled organic substrates (see review, Overbeck 1974). Attempts have been made to correlate the uptake of sulfate $-^{35}S$ to heterotrophic activity (Monheimer 1974; Jassby 1975). The use of autoradiography, although tedious, has the potential of being extremely useful in estimating the metabolic activity of bacteria (Ramsay 1974) and phytoplankton (Saunders 1972) in their natural environments. Another promising technique (Derenbach and Williams 1974) to fraction autotrophic (phytoplankton) and bacterial (heterotrophic) production by differential filtration needs to be thoroughly investigated in other aquatic ecosystems. Preliminary experiments with size fractionation by filtration in fresh water have been successful (Burnison 1975; Rudd and Hamilton 1973).

Only when the bacterial transport rates for organic substrates become known will the true role of bacteria be apparent. This will not be an easy task because of the complex bacterial–"dissolved" organic carbon–algal–detrital interrelations. Initial studies need to be made of the various techniques to measure heterotrophic functions and then the best method (or combination of methods) applied, with biomass estimates, to the Great Lakes.

In this article, the presentation of data is extensive in comparison with the degree of data interpretation. However, it is considered worthwhile to make this contribution to this study on Lake Erie because the information presented is of interest and will hopefully act as a stimulus for further research, especially in the measurement of the activity of heterotrophic bacteria.

Methods

1. Coliform Density Determinations (MF)

Coliform densities were determined on all samples by the membrane filter technique (American Public Health Association 1965). The medium used was m-Endo Agar LES. Membrane filtrations were made from appropriate volumes of each water sample. The plates were incubated at 35 C ± 0.5 C for 24 h in a saturated humidity chamber. The development of dark red colonies with a golden sheen was interpreted as direct evidence of the presence of coliform organisms. Colony counts were performed with the aid of a 10× stereomicroscope and were calculated per 100 ml.

2. Aerobic Heterotrophic Bacterial Density Determinations

a) Standard pour plate count (SPC 20 C) (Menon et al. 1967) — Viable heterotrophic bacterial populations were determined on all 1967 samples by the standard plate count procedure (SPC 20 C) at 20 C. Bacto plate count agar was the medium used. Plate counts were made using appropriate dilutions of each sample and, after being mixed with liquified (45 C) Bacto-Plate Count Agar, allowed to solidify and then incubated at 20 C for 48 h. Counts were recorded in terms of colony forming units per milliliter of water.

It is known that plate count enumeration procedures estimate variable fractions of the total bacterial population. However, using these procedures distinctly, consistent bacterial distribution patterns were observed.

b) Membrane filtration count (20 C) (Van Otterloo et al. 1968) — Viable heterotrophic bacterial populations were determined on all 1970 samples by membrane filtration method using 0.45-μm black gridded membrane filters and a modified agar, the composition of which was as follows: peptone, 3.0 g; yeast extract, 0.1 g; sodium caseinate, 0.5 g; K_2HPO_4, 0.2 g; $MgSO_4 \cdot 7H_2O$, 0.05 g; $FeCl_3$, 0.001 g; Bacto agar, 15 g; and distilled water, 1 liter. The final pH of the solution was 7.2. Incubation was at 20 C for 48 h. Counts were recorded on a per milliliter basis.

It is also known that the pour plate technique (20 C) estimates relatively larger populations for a given sample as compared to membrane filtration procedure (20 C) (B. J. Dutka Canada Centre for Inland Waters, unpublished data). Thus comparisons between the actual numbers of heterotrophs observed during 1967 and 1970 are not possible. However, the distribution patterns are comparable.

Results and Discussion

Bacterial Distribution, 1967 and 1970

a) Epilimnion aerobic heterotrophs (Viable Indigenous populations) — During the early summer of 1967 (Fig. 1) the main body of the lake, that is the offshore waters of Central and Eastern basins, had heterotrophic bacterial populations between 50 and 100 per milliliter. However, during the latter part of the summer (Fig. 2) and during the fall (Fig. 3) this area of the lake exhibited a declining trend, with concentrations of less than 50 per milliliter. The Western basin had populations consistently higher than 100 per milliliter; however, certain nearshore areas in the

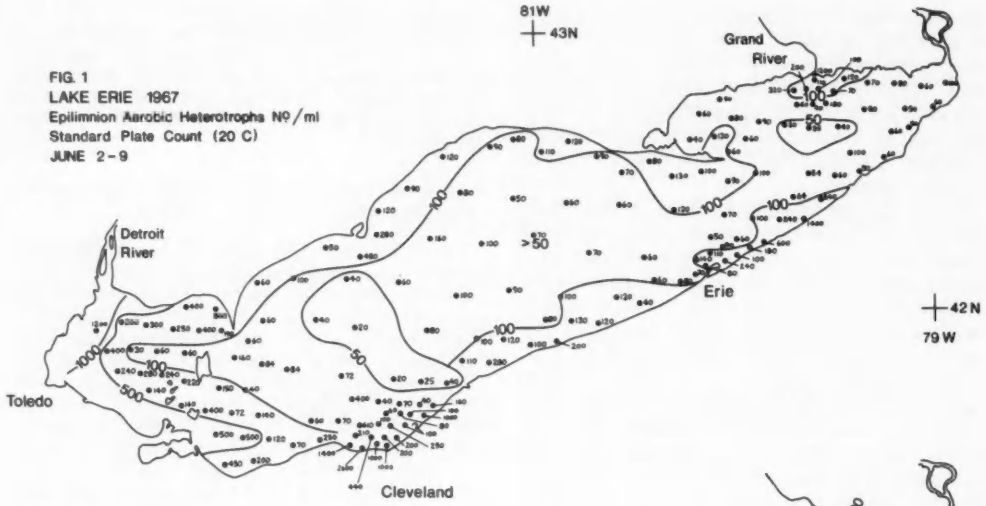

FIG. 1
LAKE ERIE 1967
Epilimnion Aerobic Heterotrophs Nº/ml
Standard Plate Count (20 C)
JUNE 2 - 9

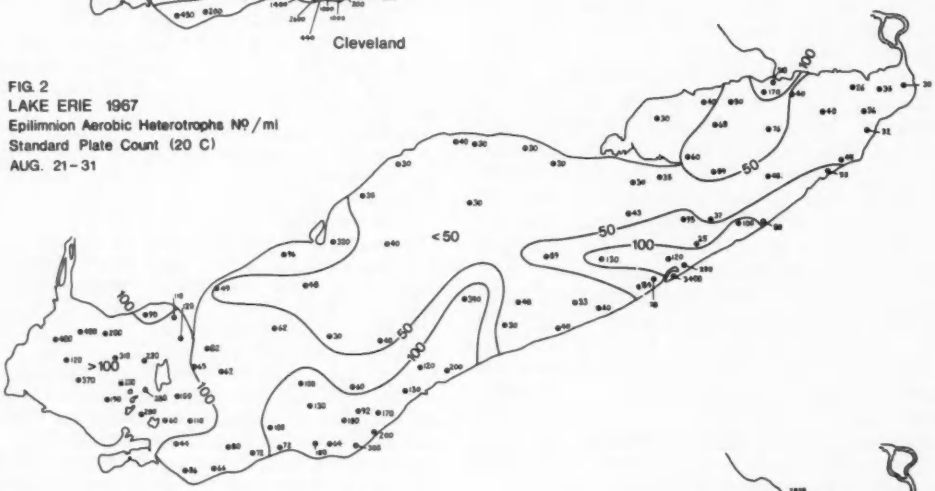

FIG. 2
LAKE ERIE 1967
Epilimnion Aerobic Heterotrophs Nº/ml
Standard Plate Count (20 C)
AUG. 21 - 31

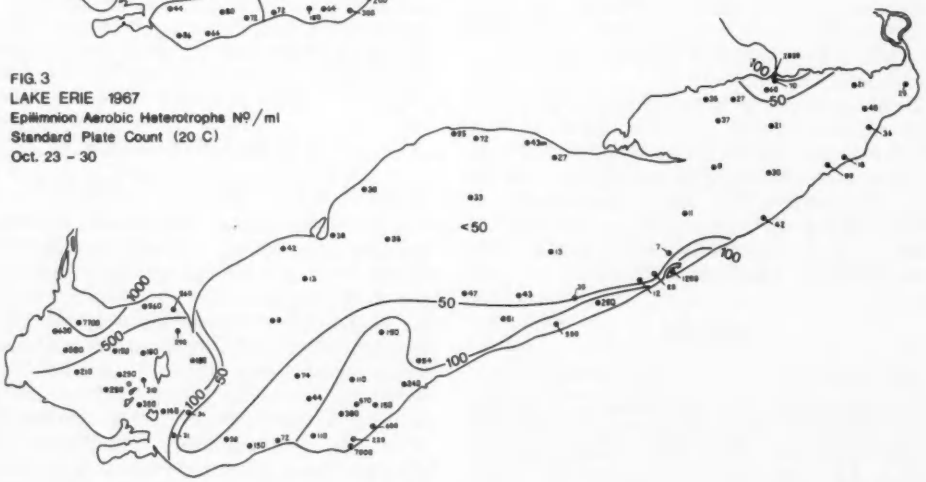

FIG. 3
LAKE ERIE 1967
Epilimnion Aerobic Heterotrophs Nº/ml
Standard Plate Count (20 C)
Oct. 23 - 30

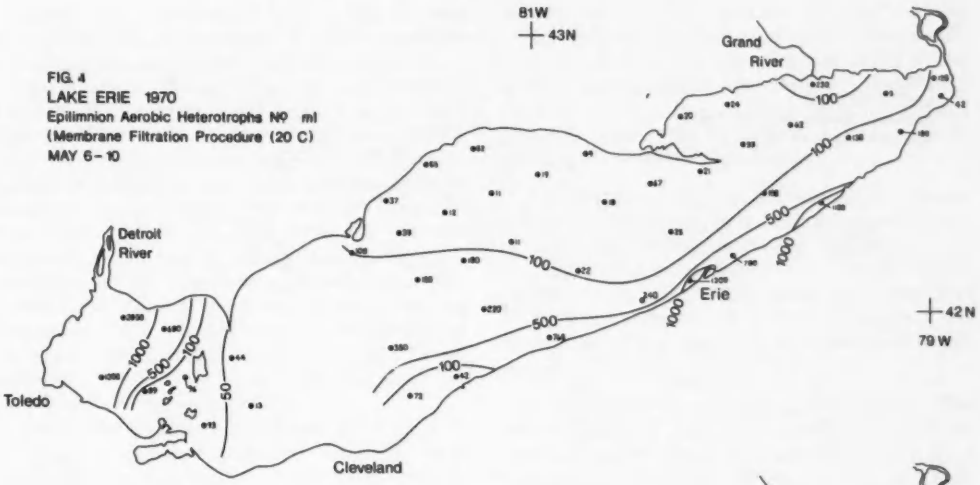

FIG. 4
LAKE ERIE 1970
Epilimnion Aerobic Heterotrophs Nº ml
(Membrane Filtration Procedure (20 C)
MAY 6 – 10

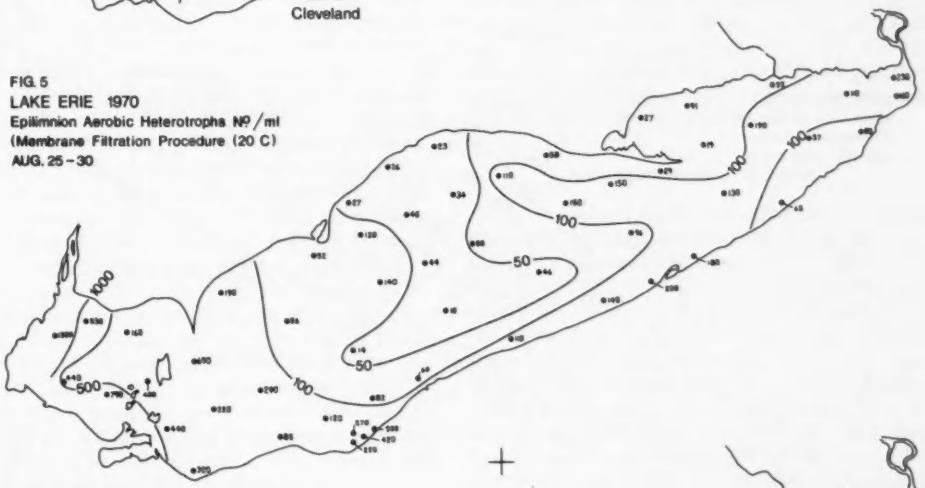

FIG. 5
LAKE ERIE 1970
Epilimnion Aerobic Heterotrophs Nº/ml
(Membrane Filtration Procedure (20 C)
AUG. 25 – 30

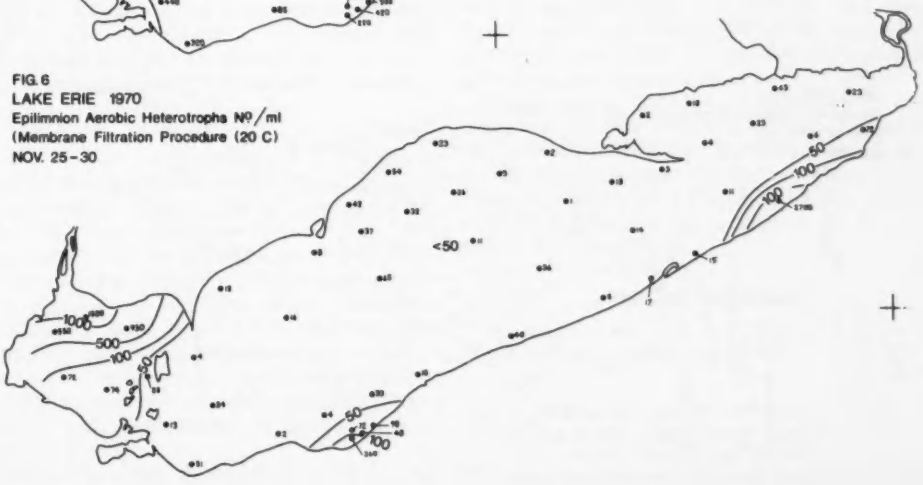

FIG. 6
LAKE ERIE 1970
Epilimnion Aerobic Heterotrophs Nº/ml
(Membrane Filtration Procedure (20 C)
NOV. 25 – 30

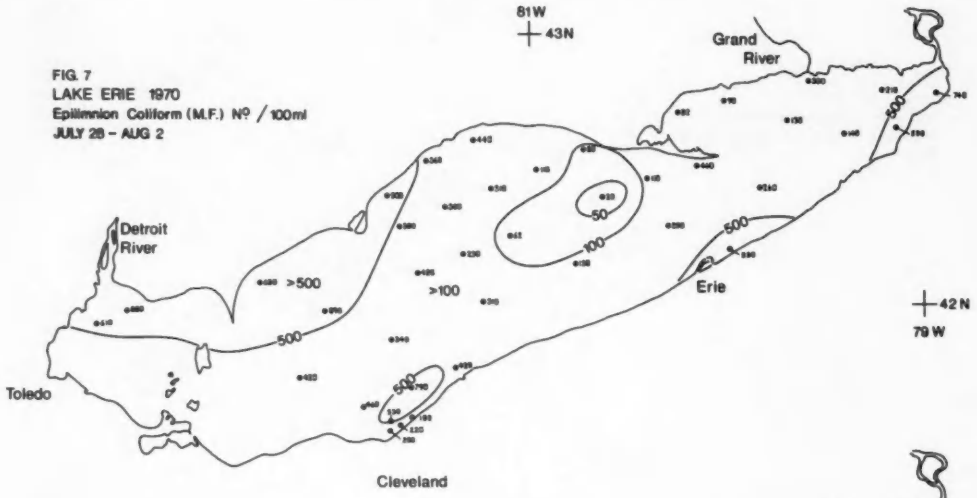

FIG. 7
LAKE ERIE 1970
Epilimnion Coliform (M.F.) Nº / 100ml
JULY 28 – AUG 2

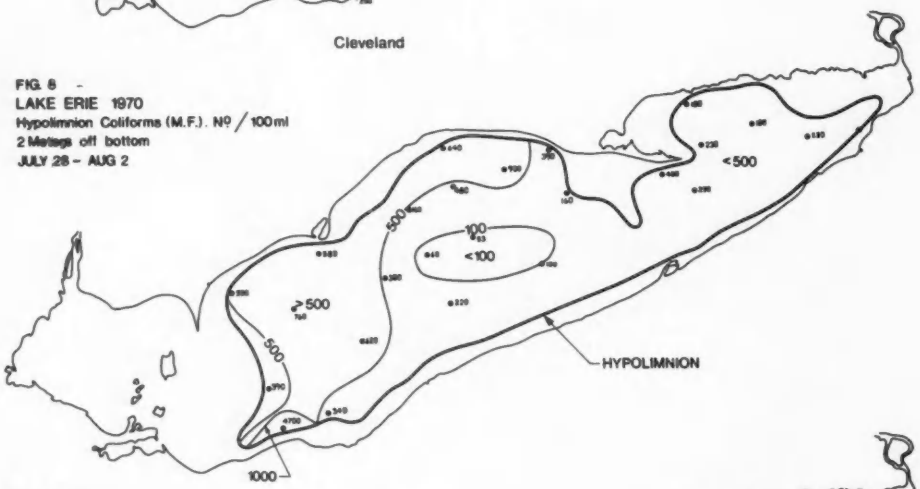

FIG. 8
LAKE ERIE 1970
Hypolimnion Coliforms (M.F.). Nº / 100ml
2 Metres off bottom
JULY 28 – AUG 2

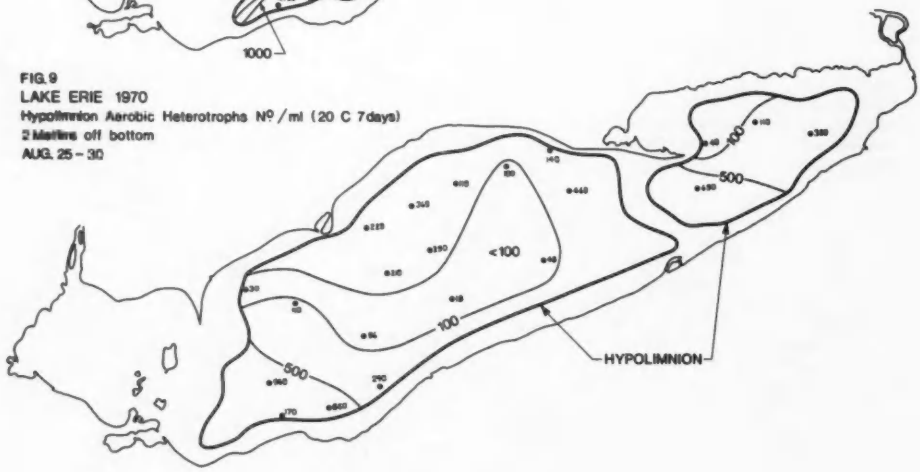

FIG. 9
LAKE ERIE 1970
Hypolimnion Aerobic Heterotrophs Nº / ml (20 C 7 days)
2 Metres off bottom
AUG. 25 – 30

Central basin (Cleveland) and in the Eastern basin (Erie and mouth of the Grand River, Ontario) also had consistently higher densities.

During 1970, spring (Fig. 4) and summer populations of heterotrophs (Fig. 5) in the main lake were low with no large difference between these two periods in terms of Central basin distribution. Nevertheless, the Eastern basin offshore values did increase from >50 to >100 per milliliter. These values are similar to the summer values of 1967 (Fig. 2), demonstrating the comparability of the distribution pattern using different bacteriological methods. However, during November (Fig. 6) there was an overall decline in densities to lake-wide value of less than 50 per milliliter. The Western basin again had the highest populations throughout 1970.

During the summer of 1967 and 1970 (Figs. 1, 2, and 5) the lake exhibited relatively large heterotrophic populations in all basins in the southern shore area. It is noteworthy that the phytoplankton inshore maximum observed in May 1970 in the Central and Eastern basins was along both north and south shores (Munawar and Munawar 1976). This inshore biomass maximum could be expected to provide substrates for subsequent heterotrophic growth. The high inshore heterotroph values observed around the entire lake in 1967 may have resulted from an earlier phytoplankton pulse in the inshore areas.

In August 1967 and 1970 the Western basin, Cleveland, and Erie had high heterotrophic bacterial counts probably due to high nutrient inputs from Detroit, Toledo, Cleveland, and Erie. Although October (Fig. 3) reflected the same nutrient input pattern, the northern half of the Central and Eastern basins showed extremely low densities (<50 per milliliter). The overall low values of less than 50 per milliliter in November probably reflects an effect of decreased temperatures.

b) Hypolimnion aeorbic heterotrophs (1970) — Generally, during summer stratification (August) the hypolimnion values (Fig. 9) appear to be greater than the epilimnion values (Fig. 5). Major portions of the Central basin had populations greater than 100 per milliliter with certain points over 500 per milliliter. Similar high bacterial densities in the hypolimnion were recorded during 1971 summer stratification (late August). These high densities appear to be dependent on the nutrients derived from the excretion of zooplankton and degradation of phytoplankton (Menon et al 1972).

Observations made during July 1970 in the hypolimnion, clearly showed an algal mass present on the lake bottom (Braidech et al. 1972), providing a large source of organic substrate for the bacteria. Also, because of the strong vertical mixing within the shallow hypolimnion (Murthy 1972), the bacteria were distributed vertically within the hypolimnion, which would result in the observed high counts.

c) Epilimnion coliforms distribution — Certain areas, e.g. Cleveland, Erie, and the northwestern part of the Central basin had higher coliform counts than the main body of the lake (>500 per 100 ml). The high counts in the northwest area of the Central basin reflect the flow patterns described by Lam and Simons (1976) for this period.

d) Hypolimnion coliform distribution — The Western part of the Central basin had coliform counts greater than 500 per 100 ml with a maximum of 4700 per 100 ml just west of Cleveland, whereas the eastern part of the Central basin had populations less than 500 per milliliter (Fig. 8). The Eastern basin had an average coliform density of 200 per 100 ml.

Summary and Conclusions

The plate count method of enumerating aerobic heterotrophic and coliform bacteria has been used for a number of surveys of Lake Erie during the years 1967, 1970. It is recognized that plate counts do not give accurate estimates of the actual numbers of bacteria present in a sample. However, these procedures do give relative estimates of the heterotrophic and coliform bacteria present in the water and provide data suitable for the construction of distribution maps. Also, these data form a baseline study for use in planning in situ experiments to measure the rates of metabolic activity of heterotrophic bacteria.

It was found that the concentrations of coliforms and aerobic heterotrophs were the highest in the Western basin and nearshore areas. In particular, the areas near the Detroit River, Grand River (Ontario), and Cuyahoga River (Cleveland, Ohio) showed high counts as did the area near Erie, Pennsylvania.

Acknowledgments

We sincerely thank Mr B. J. Dutka and Dr N. M. Burns for their valuable suggestions.

AMERICAN PUBLIC HEALTH ASSOCIATION, AMERICAN WATER WORKS ASSOCIATION, AND WATER POLLUTION CONTROL FEDERATION. 1965. Standard methods for the examination of water and wastewater. 12th ed. New York, N.Y. 769 p.

BRAIDECH, T., P. GEHRING, AND C. KLEVENO. 1972. Biological studies related to oxygen depletion and nutrient regeneration process in the Lake Erie Central Basin, p. 45–70. *In* N. M. Burns and C. Ross [ed.] Project hypo Can. Cent. Inland Waters Pap. No. 6, U.S. E.P.A. Tech. Rep. TS-05-71-208-24.

BURNISON, B. K. 1975. Microbial ATP studies. Verh. Int. Ver. Limnol. 19: 286–290.

DERENBACH, J. B., AND P. J. LEB. WILLIAMS. 1974. Autotrophic and bacterial production: fractionation of plankton populations by differential filtration of samples from the English Channel. Mar. Biol. 25: 263–269.

ISHIDA, Y., AND H. KADOTA. 1975. A comparison between viable count and direct count of bacteria in polluted sea water. Bull. Jpn. Soc. Sci. Fish. 41: 271.

JASSBY, A. D. 1975. Dark sulfate uptake and bacterial productivity in a subalpine lake. Ecology 56: 627–636.

LAM, D. C. L., AND T. J. SIMONS. 1976. Numerical computations of advective and diffusive transports of chloride in Lake Erie, 1970. J. Fish. Res. Board Can. 33: 537–549.

MENON, A. S., W. S. GLOOSCHENKO, AND N. M. BURNS. 1972. Bacteria–phytoplankton relationships in Lake Erie. Proc. Conf. Great Lakes Res. 15: 94–101.

MENON, A. S., A. JURKOVIC, W. WINTER, AND B. J. DUTKA. 1967. Bacteriological study of Lake Erie conducted for the Advisory Board on Water Pollution, IJC, 1967. MS Rep. No. 67-19.

MONHEIMER, R. H. 1974. Sulfate uptake as a measure of planktonic microbial production in freshwater ecosystems. Can. J. Microbiol. 20: 825–831.

MUNAWAR, M., AND I. F. MUNAWAR. 1976. Lakewide study of phytoplankton biomass and its species composition in Lake Erie, April–December 1970. J. Fish. Res. Board Can. 33: 581–600.

MURTHY, C. R. 1972. An investigation of diffusion characteristics of the hypolimnion of Lake Erie, p. 39–44. *In* N. M. Burns and C. Ross [ed.] Project hypo. Can. Cent. Inland Waters Pap. No. 6; U.S. E.P.A. Tech. Rep. TS-05-71-208-24.

OVERBECK, J. 1974. Microbiology and biochemistry. Mitt. Int. Ver. Limnol. 20: 198–228.

RAMSAY, A. J. 1974. The use of autoradiography to determine the proportion of bacteria metabolizing in an aquatic habitat. J. Gen. Microbiol. 80: 363–373.

RUDD, J. W. M., AND R. D. HAMILTON. 1973. Measurement of adenosine triphosphate (ATP) in two Precambrian Shield lakes of northwestern Ontario. J. Fish. Res. Board Can. 30: 1537–1546.

SAUNDERS, G. W. 1972. Potential heterotrophy in a natural population of *Oscillatoria agardhii* var. *isothrix* Skija. Limnol. Oceanogr. 17: 204–211.

SCHMIDT, E. L. 1973. The traditional plate count technique among modern methods, p. 453–454. *In* T. Rosswall [ed.] Modern methods in the study of microbial ecology. NFR[17], Rotobeckman, Stockholm.

VAN OTTERLOO, H. R., P. COLLINS, AND B. J. DUTKA. 1968. Bacteriological Study of Lake Ontario, Lake Erie, Lake Huron and Lake Superior; conducted for the Advisory Board on Water Pollution, IJC 1968. MS Rep. No. KR-68-3.

A Lakewide Study of Phytoplankton Biomass and its Species Composition in Lake Erie, April–December 1970[1]

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MUNAWAR, M., AND I. F. MUNAWAR. 1976. A lakewide study of phytoplankton biomass and its species composition in Lake Erie, April–December 1970. J. Fish. Res. Board Can. 33: 581–600.

Surface samples of phytoplankton were collected from April to December 1970 at approximately 4-wk intervals from 25 stations distributed over the entire lake. Taxonomic identification and enumeration was done by the Utermöhl technique. About 125–150 species were identified in each basin. "Less common" or rare species contributed significantly to the total phytoplankton biomass. The species comparison among the three basins indicated that species like *Rhodomonas minuta* and *Cryptomonas erosa* were perennial in all the basins. *Aphanizomenon flos-aquae, Anabaena spiroides* var. *contracta, Coscinodiscus rothii, Actinastrum hantzschii,* and *Ceratium hirundinella* were typical for the Western basin. *Ochromonas* spp. and *Peridinium aciculiferum* were common in the Central basin whereas *Gymnodinium helveticum* and *G. uberrimum* were characteristic for the Eastern basin.

The diatoms contributed least in the Eastern basin (36%) but showed high percentage in both Central (55%) and Western (58%) basins. The phytoflagellates made important contributions in the three basins; highest percentage (41%) was in the Eastern basin. Phytoflagellate abundance increased from west to east and appeared to have a wide range of adaptability to different nutrient conditions. During July and August high biomass concentration but uneven horizontal distribution was observed. Phytoplankton biomass ranged between 0.8 and 13.2, 0.6 and 6.0, and 1.0 and 4.2 g/m^3 in the Western, Central, and Eastern basins, respectively. The highest biomass of 13.2 g/m^3 was in the Western basin during April. Inshore–offshore differentiation was observed in the Central and Eastern basins with many peaks in the inshore region. Size analysis of phytoplankton on a few stations showed that the nannoplankton ($\leq$64 μ) comprised a major portion of phytoplankton biomass during spring and fall periods. Based on maximum biomass concentration the Lake Erie Western basin was classified as highly eutrophic, the Eastern basin as mesotrophic, and the Central basin between the mesotrophic and eutrophic conditions.

MUNAWAR, M., AND I. F. MUNAWAR. 1976. A lakewide study of phytoplankton biomass and its species composition in Lake Erie, April–December 1970. J. Fish. Res. Board Can. 33: 581–600.

Des échantillons de surface de phytoplancton ont été recueillis d'avril à décembre 1970 à intervalles d'environ 4 semaines à 25 stations réparties sur tout le lac. L'identification taxonomique et le dénombrement ont été faits selon la méthode d'Utermöhl. Environ 125–150 espèces ont été identifiées dans chaque bassin. Les espèces rares ou « moins communes » représentent une portion significative de la biomasse totale du phytoplancton. Une comparaison des espèces des trois bassins indique que des espèces telles que *Rhodomonas minuta* et *Cryptomonas erosa* sont toujours présentes dans tous les bassins. *Aphanizomenon flos-aquae, Anabaena spiroides* var. *contracta, Coscinodiscus rothii, Actinastrum hantzschii* et *Ceratium hirundinella* sont typiques du bassin occidental. *Ochromonas* spp. et *Peridinium aciculiferum* sont communs dans le bassin central, alors que *Gymnodinium helveticum* et *G. uberrimum* sont caractéristiques de bassin oriental.

La contribution des diatomées est plus faible dans le bassin oriental (36%), mais représente un pourcentage élevé tant dans le bassin central (55%) que dans le bassin occidental (58%). Les phytoflagellés apportent des contributions importantes dans les trois bassins; le pourcentage le plus élevé (41%) se trouve dans le bassin oriental. L'abondance de phytoflagellés augmentent d'ouest en est et semblent pouvoir s'adapter à une gamme étendue de conditions nutritives différentes. En juillet et août, on observe une haute concentration de biomasse mais une répartition horizontale inégale. La biomasse phytoplanctonique varie de 0.8 et 13.2, 0.6 et 6.0 et

[1]Paper presented at the 36th annual meeting of the American Society of Limnology and Oceanography, Salt Lake City, Utah, USA 1973.

Printed in Canada (J4122)
Imprimé au Canada (J4122)

1.0 et 4.2 g/m³ dans les bassins occidental, central et oriental respectivement. La biomasse la plus élevée, 13.2 g/m³, se trouve dans le bassin occidental en avril. Nous avons observé des différences entre le littoral et le large dans les bassins central et oriental, la région littorale présentant plusieurs pics. L'analyse des tailles du phytoplancton à quelques stations démontre que le nannoplancton (64 μ) constitue une portion majeure de la biomasse du phytoplancton au printemps et en automne. Sur la base de concentration maximale de biomasse, le bassin occidental du lac Erié est classé comme hautement eutrophe, le bassin oriental comme mésotrophe, alors que le bassin central est intermédiaire entre des conditions mésotrophes et eutrophes.

Accepted November 18, 1975 Accepté le 18 novembre 1975

LAKEWIDE, comprehensive studies on Lake Erie phytoplankton are few in number (Davis 1966, 1969; Vollenweider et al. 1974; Munawar and Munawar 1975). The phytoplankton of western Lake Erie has received the attention of several investigators (Tiffany 1934; Chandler 1940, 1942, 1944; Verduin 1951, 1952, 1960, 1964; Wright et al. 1955; Casper 1965; Wujeck 1967; Hohn 1969; Olive et al. 1969; and Taft and Taft 1971). Studies of the Central basin have been confined to a few stations mainly in nearshore waters (Burkholder 1929a, 1929b, 1960; Davis 1954a, 1954b, 1964, 1965; Braidech et al. 1971). Not much is known about eastern Lake Erie phytoplankton except for a few older reports by Kellicot 1878; Mills 1882; Burkholder 1929a, 1929b, 1960; Gottschall 1930; and Gottschall and Jennings 1933. These past studies have suffered from a number of limitations which are:

a) No lakewide studies were made, resulting only in local and regional information;

b) The investigations were usually extended over a short time period;

c) Different techniques of preservation and concentration were used by various investigators, thus producing data that are not comparable;

d) Much emphasis was given to netplankton, with nannoplankton being neglected, which provided incomplete information about the total phytoplankton community. As a consequence, taxonomic groups other than diatoms and bluegreens did not receive attention, resulting in an underestimation of phytoflagellates (Chrsomonadinae, Cryptomonadinae, and Dinophycinae) and microalgae in Erie phytoplankton;

e) Generally, only "common" species (those found frequently in the sample) were counted and the "less common" or rare species were neglected. The enumeration procedure needs particular attention since it has already led to significant underestimation of the total phytoplankton biomass;

f) The abundance data were reported as cell numbers and were not converted to cell volume or biomass. Such data are not suitable for use in ecological research since phytoplankton show a great degree of diversity in size.

A research program was initiated at the Fisheries Research Board of Canada detachment, Canada Centre for Inland Waters, Burlington during 1970 which attempted to remedy these deficiencies and obtain adequate information about the phytoplankton biomass and its species composition and the results are presented here.

Materials and Methods

Lakewide samples were collected at approximately 4-wk intervals from 25 stations from April to December 1970 (Fig. 1). The offshore stations are defined as those that are more than 18 m in depth.

Samples were collected by Van Dorn bottle from 1- and 5-m depth, mixed together in equal parts, and preserved with modified Lugol's solution (Nauwerck 1963; Vollenweider 1969). Another subsample was kept alive in a container maintained at the surface temperature of the lake. These live samples were used to identify living organisms, particularly the motile phytoflagellates.

The taxonomic identification and enumeration was done using an inverted microscope (Utermöhl 1958). Depending on phytoplankton population density, aliquots of 5–25 ml were sedimented in settling chambers and examined with the inverted microscope (Model: Wild Heerbrugg M40, phase contrast).

The enumeration procedure developed is briefly described below:

1) The settled material was scanned and "common" netplankton species were counted in two transects under 300× magnification;

2) "Less common" and rare species of netplankton were enumerated in the entire chamber under 300× magnification;

3) The nannoplankton and microalgae were counted in two transects under 600× magnification.

In each sample, at least 300 units or entities were counted (Lund et al. 1958; Nauwerck 1963) where each colony was treated as a unit. Diatoms were identified from permanent slides prepared by heating the sample in a drop or two of H_2O_2 on a bunsen flame and mounted in Hyrax medium. The diatoms were identified with Carl Zeis Standard Universal Microscope equipped with Nomarski differential interference contrast illumination.

The cell volume of each species was computed from the average cell dimensions measured in each sample and the geometrical shape that most closely simulated the shape of the species (Vollenweider

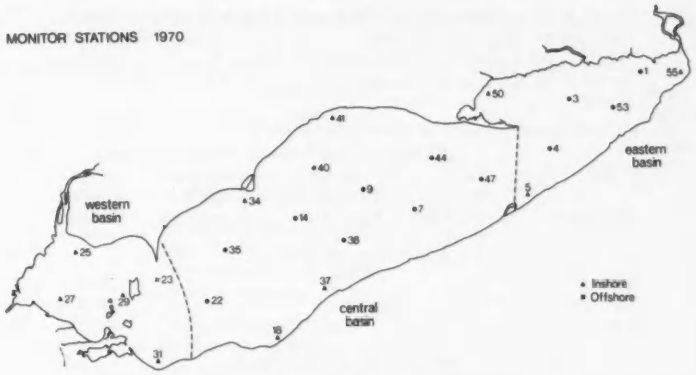

FIG. 1. Stations sampled in the 1970 lakewide survey.

1969). In the case of colonial forms, the average number of cells per colony were determined. These were then multiplied by the computed volume per cell to obtain the volume per colony. The cell volume was converted to biomass (mg/m³) assuming the specific gravity of phytoplankton to be 1.0 (Willen 1959; Nauwerck 1963). While the relative merits of cell volume determinations are accepted and routinely used by algologists, extra care is required to make these calculations. The accuracy of cell volume determinations depends on the accuracy of the measurement of cell dimensions. In the present study, at least 25 individuals were measured for each "common" species in each sample and no standard cell volumes were used so as to take the wide range of size diversity observed into account. Although the cell volume procedure is time-consuming and not without problems, it is preferable to expressing the abundance in a mixed population as cell numbers. The cell volume technique provides the total phytoplankton biomass by species composition while excluding zooplankton and detrital biomass. These biomass estimates can also be converted to carbon or nitrogen equivalents to compute conversion of nutrients to biomass or the reverse.

Based on the average longest dimension in each sample, the species greater than 64 μ were classified as netplankton with nannoplankton being equal to or less than 64 μ. The weighted mean biomass concentration in the lake was obtained by using the weighting factors of 15.6, 59.6, and 24.6% with the biomass concentrations in each survey for the Western, Central, and Eastern basins, respectively. The three surveys conducted between April 7 and June 6 have been considered as spring surveys, the four between July 3 and September 27 as summer surveys, and the three between October 21 and December 18 as fall surveys. There were no winter surveys.

The data for the total chlorophyll *a* (uncorrected) and primary production have been obtained from Glooschenko et al. (1974a, b) who have described the methods followed.

Results and Discussion

In the past lakewide information about species composition in Lake Erie has been inadequate. Realistic analyses of phytoplankton biomass composition was limited by the analytical technique and enumeration procedures (Munawar et al. 1974). The analytical technique (Utermöhl) used in the present study is a standard and universally accepted procedure. Both "common" and "less common" species have been enumerated.

The enumeration procedure can have a great impact on the biomass data. Normally, due to the time-consuming nature of the taxonomic work, only "common" species are counted and since these "common" species are few, they can be counted in a relatively short time. Although the enumeration time is considerably reduced by such a procedure, it neither provides information about the total phytoplankton community nor anything about the "less common" or rare species. These "less common" species may contribute significantly to the total phytoplankton biomass, and their abundance or disappearance might provide important information in assessing the long-range changes in species composition due to eutrophication. The collective importance of the "less common" species is demonstrated in Table 1 where a few examples from the three basins are given. This clearly indicates the need that the enumeration procedure must give due consideration to both "common" and rare species as was done in the present study.

SPECIES COMPOSITION

The detailed listing of the species of phytoplankton (both net plankton and nannoplankton)

TABLE 1. A few examples of the percent contribution of "common" and "less common" species in Lake Erie basins during 1970. Station No. 1 is off Colborne (Eastern basin), Station No. 7 is a midlake station (Central basin), and Station No. 23 is off Bass Island (Western basin).

	Common		Less common	
Examples	Percent cell numbers	Percent biomass	Percent cell numbers	Percent biomass
	Eastern basin Station No. 1			
Spring	95.0	13.0	5.0	87.0
Summer	93.0	20.0	7.0	80 0
Fall	82.0	1.0	18.0	99.0
	Central basin Station No. 7			
Spring	94.5	3.0	5.5	97.0
Summer	92.0	10.0	8.0	90.0
Fall	86.0	21.0	14.0	79.0
	Western basin Station No. 23			
Spring	89.0	32.0	11.0	68.0
Summer	90.0	10.0	10.0	90.0
Fall	60.0	2.0	40.0	98.0

observed in Lake Erie is on the file at Canada Centre for Inland Waters.

From 125 to 150 taxa were identified in the Lake Erie basins (Table 2). Large number of species of Chlorophyta, Diatomeae, and Cyanophyta were present in decreasing order of species numbers. More than 70% of the total number of species identified belonged to Chlorophyta and phytoflagellates (Chrysomonadinae, Cryptomonadinae, and Dinophycinae). The Chlorophyta showed the highest number of species. It is interesting to compare these data with those of Tiffany (1937) and Taft and Taft (1971) who also reported highest number of species in western Lake Erie were of the Chlorophyta group indicating the continued dominance of this group for the past 30 yr. The phytoflagellates showed a 13, 12, and 18% contribution to species numbers in the Western, Central and Eastern basins, respectively.

SEASONAL SPECIES SUCCESSION

The seasonal abundance of some of the frequently found species is shown in Fig. 2, 3, and 4 for the three basins. Tables 3, 4, and 5 provide a list of common species (those contributing 5% or more to the total biomass) by survey. These data permit the following main features to be outlined:

In the Western basin the most significant contribution to the species succession was made by *Aphanizomenon flos-aquae* (Aug.–Oct.), *Conscinodiscus rothii* (July–Nov.), *Fragilaria crotonensis* (April), *Stephanodiscus tenuis* (April–May), and *Cryptomonas erosa* (May, July, August). The perennial species were: *Stephanodiscus tenuis, Melosira islandica* spp. *helvetica, M. binderana, Fragilaria crotonensis, Rhodomonas minuta*, and *Cryptomonas erosa*. The most significant contributions to the species composition in the Central basin was by *Fragilaria crotonensis* (July, August), *F. capucina* (Sept.), *Stephanodiscus niagarae* (May, Sept., Dec.), and *Ceratium hirundinella* (Aug., Sept.). The perennial species in this basin were: *Ochromonas* spp., *Stephanodiscus niagarae, Rhodomonas minuta*, and *Cryptomonas erosa*.

The seasonal succession of phytoplankton in

TABLE 2. Number of species identified and percentage of species belonging to various taxonomic groups.

	Western basin	Central basin	Eastern basin
No. of species	144	125	134
	% composition		
Cyanophyta	7	12	15
Chlorophyta	61	60	53
Euglenophyta	1	0	1
Chrysomonadinae	6	5	8
Diatomeae	19	16	14
Cryptomonadinae	3	3	4
Dinophycinae	3	4	5

TABLE 3. Seasonal abundance of the most common species that contributed 5% or more to the total phytoplankton biomass in the Western basin of Lake Erie.

Months	Western basin species	Mean % of total vol.
Apr. 7–11	*Fragilaria crotonensis* (Edw.) Kitton	62.5
	Stephanodiscus tenuis Hust.	17.0
	Melosira binderana Kg.	8.5
May 6–10	*Cryptomonas erosa* Ehrnb.	20.5
	Stephanodiscus tenuis	14.5
	Rhodomonas minuta Skuja	9.0
	Melosira binderana	7.5
June 2–6	*Cryptomonas erosa*	18.0
	Rhodomonas minuta	14.0
	Gymnodinium helveticum Penard	12.0
	Staurastrum paradoxum (Meyen)	8.5
	Stephanodiscus tenuis	7.5
July 3–7	*Coscinodiscus rothii* (E.) Grün.	23.0
	Cryptomonas erosa	22.0
	Stephanodiscus niagarae E.	9.5
	Rhodomonas minuta	8.0
	Aphanizomenon flos-aquae (L.) Ralfs	9.0
July 25–Aug. 2	*Ceratium hirundinella* (O. Müll.) Schrank	20.5
	Pediastrum simplex (Meyen) Lemm.	19.5
	Aphanizomenon flos-aquae	13.0
	Stephanodiscus niagarae	10.0
	Coscinodiscus rothii	9.0
	Cryptomonas erosa	7.5
Aug. 25–30	*Coscinodiscus rothii*	35.5
	Aphanizomenon flos-aquae	14.0
	Cryptomonas erosa	12.5
	Rhodomonas minuta	10.0
Sept. 23–27	*Coscinodiscus rothii*	27.0
	Aphanizomenon flos-aquae	20.0
	Rhodomonas minuta	13.5
	Stephanodiscus tenuis	6.5
	Cryptomonas erosa	6.5
Oct. 21–26	*Coscinodiscus rothii*	40.5
	Melosira islandica O. Müll.	27.5
Nov. 25–30	*Stephanodiscus niagarae*	18.5
	Stephanodiscus tenuis	20.0
	Coscinodiscus rothii	14.0
	Melosira islandica	10.5
	Cryptomonas erosa	6.5
Dec. 14–18	*Stephanodiscus tenuis*	28.0
	Stephanodiscus niagarae	16.0
	Fragilaria crotonensis (Edw.) Kitton	10.0
	Melosira islandica	8.5

the Eastern basin was dominated by *Stephanodiscus tenuis* (April), *S. Niagarae* (Nov.–Dec.), *Fragilaria crotonensis* (Sept.–Oct.), *Rhodomonas minuta* (June), *Ceratium hirundinella* (July, Sept.), and *Peridinium aciculiferum* (May, June). The perennial species were: *Pediastrum dulpex*, *Fragilaria crotonensis*, *Rhodomonas minuta*, and *Cryptomonas erosa*.

The data in Tables 3, 4, and 5 on the common species demonstrate that species composition in the three basins was not very different. However, certain characteristic differences were apparent in the relative abundances and are summarized below.

In the Western basin *Aphanizomenon flos-aquae* Ralfs was abundant but it was low in con-

TABLE 4. Seasonal abundance of the most common species that contributed 5% or more to the total phytoplankton biomass in the Central basin of Lake Erie.

Months	Central basin species	Mean % of total vol.
Apr. 7–11	*Stephanodiscus tenuis* Hust.	34.0
	Rhodomonas minuta Skuja	8.0
	Chlorella sp.	20.5
	Stephanodiscus hantzschii var. *pusilla* Grün.	12.5
	Chlamydomonas globosa Snow	7.0
	Melosira binderana Kg.	5.0
May 6–10	*Stephanodiscus niagarae* E.	30.0
	Peridinium aciculiferum (Lemm.) Lemm.	17.0
	Rhodomonas minuta	12.0
	Gymnodinium helveticum Penard	7.0
	Gymnodinium uberrimum (Allman) Kofoid & Sweezy	7.5
	Cryptomonas erosa Ehrnb.	7.0
June 2–6	*Gymnodinium helveticum*	24.0
	Rhodomonas minuta	23.0
	Cryptomonas erosa	13.5
	Chrysochromulina parva Lackey	6.0
	Chroococcus dispersus var. *minor* G. M. Smith	5.5
	Gymnodinium uberrimum	5.0
July 3–7	*Cryptomonas erosa*	34.0
	Fragilaria crotonensis (Edw.) Kitton	32.5
	Rhodomonas minuta	15.0
25 July–2 Aug.	*Ceratium hirundinella* (O. Müll.) Schrank	37.5
	Fragilaria crotonensis	13.0
	Anabaena spiroides Klebahn	12.5
	Oedogonium sp.	10.0
	Aphanizomenon flos-aquae (L.) Ralfs	8.5
Aug. 25–30	*Ceratium hirundinella*	31.0
	Staurastrum paradoxum (Meyen)	17.0
	Fragilaria crotonensis	15.0
Sept. 23–27	*Fragilaria capucina* (Desm.)	37.0
	Stephanodiscus niagarae E.	11.0
	Ceratium hirundinella	8.5
	Fragilaria crotonensis	7.5
Oct. 21–26	*Stephanodiscus niagarae*	54.0
	Rhodomonas minuta	9.5
	Cryptomonas erosa	6.5
Nov. 25–30	*Stephanodiscus niagarae*	71.0
	Pediastrum simplex (Meyen) Lemm.	6.5
Dec. 14–18	*Stephanodiscus niagarae*	73.5
	Pediastrum simplex	8.5

centration in the Eastern and Central basins. *Anabaena spiroides* var. *contracta* Kleb. was well developed in both the Western and Central basins (more than in the Central basin) but did not contribute significantly in the Eastern basin. Certain species were characteristic in each basin; e.g. *Coscinodiscus rothii* (E) Grun, *Actinastrum hantzschii* lagerh., and *Ceratium hirundinella* (O. F. Muller) Schrank, were typical for the Western basin. Species of *Ochromonas* and the dinoflagellate *Peridinium aciculiferum* Lemm. were most common in the Central basin. The Eastern basin was characterized by abundance of species of *Gymnodinium*, *Glenodinium*, and *Amphidinium*, particularly *Gymnodinium helveticum* Penard and *G. uberrimum* (Allman) Kofoid and Swezy.

The species composition of western Lake Erie phytoplankton is the best known of the Great Lakes phytoplankton, although past studies have

TABLE 5. Seasonal abundance of the most common species that contributed 5% or more to the total phytoplankton biomass in the Eastern basin of Lake Erie.

Months	Eastern basin species	Mean % of total vol.
Apr. 7–11	*Stephanodiscus tenuis* Hust.	85.0
May 6–10	*Peridinium aciculiferum* (Lemm.) Lemm	27.0
	Chroococcus dispersus var. *minor* G. M. Smith	13.0
	Stephanodiscus niagarae E.	9.0
	Gymnodinium uberrimum (Allman) Kofoid & Sweezy	8.0
	Cryptomonas erosa Ehrnb.	7.0
June 2–6	*Rhodomonas minuta* Skuja	37.5
	Gymnodinium helveticum	11.0
	Ochromonas sp.	12.5
	Cryptomonas erosa	10.0
	Chroococcus dispersus var. *minor*	6.0
July 3–7	*Rhodomonas minuta*	34.5
	Cryptomonas erosa	20.0
	Chroococcus dispersus var. *minor*	11.5
	Chrysochromulina parva Lackey	10.0
25 July–2 Aug.	*Ceratium hirundinella* (O. Müll.) Schrank	25.0
	Aphanizomenon flos-aquae (L.) Ralfs	17.0
	Oedogonium sp.	11.5
	Rhodomonas minuta	11.5
	Cryptomonas erosa	6.0
Aug. 25–30	*Rhodomonas minuta*	14.0
	Ceratium hirundinella	12.5
	Cosmarium sp.	13.0
	Pediastrum simplex (Meyen) Lemm.	7.0
	Sphaerocystis sp.	7.5
Sept. 23–27	*Ceratium hirundinella*	22.0
	Fragilaria crotonensis (Edw.) Kitton	18.0
	Rhodomonas minuta	18.5
	Cosmarium sp.	10.5
	Staurastrum paradoxum (Meyen)	7.0
Oct. 21–26	*Cryptomonas erosa*	15.0
	Fragilaria crotonensis	11.5
	Fragilaria capucina (Desm.)	10.0
	Pediastrum simplex	8.0
	Staurastrum paradoxum	8.5
	Rhodomonas minuta	9.0
	Stephanodiscus niagarae	10.0
Nov. 25–30	*Stephanodiscus niagarae*	64.5
	Chlamydomonas globosa Snow	7.0
Dec. 14–18	*Stephanodiscus niagarae*	80.0

been restricted to a few taxonomic groups. However, they provide an interesting comparison with this study and will update the species composition of diatoms since it is believed that the floristic changes in western Lake Erie have paralleled eutrophication trends (Verduin 1964; Hohn 1969). For example Verduin (1964) has shown some community changes in western Lake Erie during 1948–62. He pointed out that before 1950 the dominant species were *Asterionella formosa*, *Tabellaria fenestrata*, *Melosira ambigua*, whereas during 1960–61 the community was dominated by *Fragilaria capucina*, *Coscinodiscus radiatus*, and *Melosira binderana*. This comparison more or less confirms Verduin's findings in that those species that were dominant before 1950 (*A. formosa*, *T. fenestrata*, and *M. ambigua*) continued to be less important in the 1970 collections. A comparison with the diatoms found in 1961–62 shows that *Coscinodiscus rothii* and

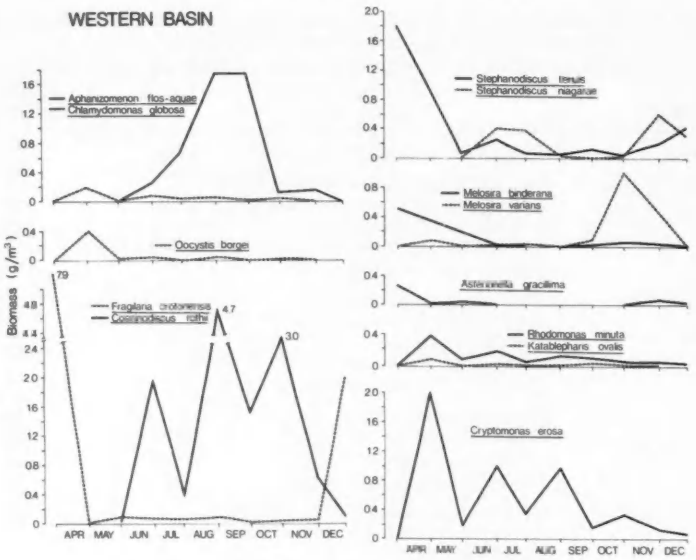

FIG. 2. Species succession in Western Lake Erie, 1970.

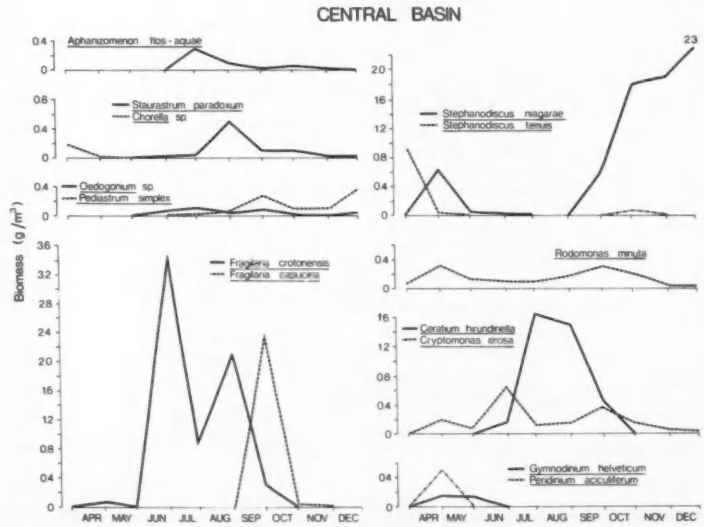

FIG. 3. Species succession in Central Lake Erie, 1970.

Melosira binderana continue to dominate for almost 8 yr but *F. capucina*, although quite common in our collections, is not important when its percent contribution to the total biomass is considered (Table 6).

Another comparison can be made with the data of Hohn (1969) on the long-term changes of diatoms over the past several years based on analysis of data from 1938 to 1940, 1948 to 1949 and 1960 to 1965. He drew a number of conclusions: 1) Species that were present previously in large numbers had almost disappeared (e.g. *Cyclo-*

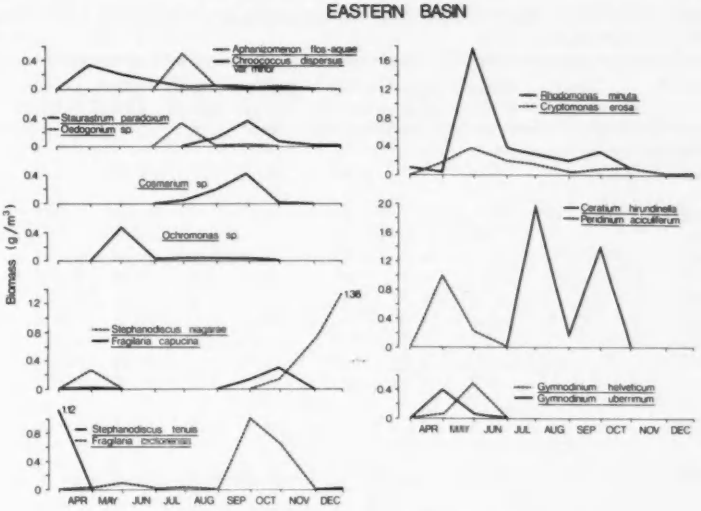

FIG. 4. Species succession in Eastern Lake Erie, 1970.

tella stelligera and *Rhizosolenia eriensis*); 2) *Tabellaria fenestrata* has lost its relative importance in the species complex; 3) Species that were not present or observed only occasionally have become major dominants: *Melosira binderana, Diatoma tenue* var. *elongatum, Cyclotella meneghiniana, Stephanodiscus alpinus*, and *S. tenuis*; 4) *Fragilaria crotonensis* has shown significant increase. *Fragilaria capucina* has increased in abundance more than any other species during recent years; 5) *Coscinodiscus radiatus* increased considerably since 1938–40 collections.

Our data on common (Table 3), perennial (Fig. 2), and less common species (Table 6) are in general agreement with the findings of Verduin (1964) and Hohn (1969) concerning the diatom species composition in the Western basin. Furthermore, some difference of taxonomic opinion about certain species is apparent (e.g. *Coscinodiscus radiatus* is identified as *C. rothii* in the present study). Some of the comparative findings are:

a) Species such as *Cyclotella stelligera* and *Rhizosolenia eriensis* were rarely recorded. *Asterionella formosa* and *Tabellaria fenestrata*'s contribution was insignificant and were unimportant relatively. This confirms the earlier studies;

b) Eutrophic indicator species *Melosira binderana* and *Stephanodiscus* spp. were frequent (*S. Niagarae*, *S. tenuis*, and *S. hantzschii*) whereas Hohn mentions *S. alpinus* to be dominant which was not observed in the present study. Similarly Verduin mentions *Fragilaria capucina* as dominant in 1960–61, whereas in the 1970 collections although it was found quite often in the samples but it was not dominant;

c) Such eurytopic species like *Fragilaria crotonensis* have remained abundant;

d) *Coscinodiscus rothii* has remained abundant;

e) In addition, it was observed that *Cryptomomas erosa* and *Rhodomonas minuta* were common in all the samples;

f) Eutrophic species like *Aphanizomenon flos-aquae* and *Anabaena spiroides* were frequent during summer in the Western basin.

THE MEAN GROUP COMPOSITION

The mean group composition on a lakewide basis for various stations is shown in Fig. 5. In general, the Western basin stations possessed the highest percentage of diatoms whereas the Eastern basin had the lowest percentage. The dinoflagellates constituted a relatively high percentage of the algae in the Eastern basin whereas the highest percentage of greens was observed near the mouth of the Detroit River. It is quite likely that an earlier peak of diatoms may have occurred and then been missed as was indicated by numerous empty frustules in early April collections in the Western basin. Figure 5 indicates that phytoflagellates (Chrysomonadinae, Cryptomonadinae,

TABLE 6. Occurrence of "less common" diatom species that contributed less than 5% to the total phytoplankton biomass in western Lake Erie during 1970.

Species	Apr. 7–11	May 6–10	June 2–6	July 3–7	25 July–2 Aug.	Aug. 25–30	Sept. 23–27	Oct. 21–26	Nov. 25–30	Dec. 14–18
Asterionella formosa		×	×	×	×			×		×
A. gracillima		×		×	×		×	×	×	×
Cocconeis sp.										×
Coscinodiscus rothii				×	×		×			
Diatoma elongatum			×							
Fragilaria capucina		×	×	×			×	×	×	
F. crotonensis		×	×	×	×	×	×	×	×	
Melosira binderana		×	×	×				×		×
M. varians		×	×	×	×		×	×	×	
Navicula sp.										×
Nitzschia sp.							×			
N. dissipata				×						
N. linearis										×
N. palea							×	×	×	×
Rhizosolenia sp.										×
Stephanodiscus hantzschii	×	×	×	×	×	×	×	×	×	×
S. niagarae				×	×	×		×		
S. tenuis			×		×	×		×	×	
Surirella angustata								×		
Surirella sp.										×
Synedra sp.			×				×			
S. ulna			×	×		×			×	×
S. utermohlii				×						
Tabellaria fenestrata		×	×							
T. flocculosa				×	×		×			×

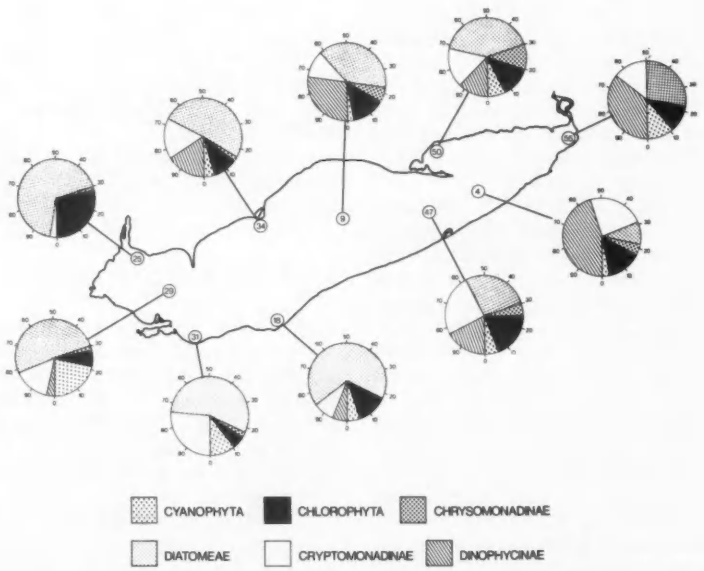

FIG. 5. The phytoplankton group composition for a few stations in Lake Erie where the percentage represents mean of 10 surveys from April–December, 1970.

TABLE 7. Mean values of phytoplankton biomass and its percentage composition for various seasons and for the 10 surveys in Lake Erie basins during 1970.

	Eastern basin				Central basin				Western basin			
	Spring	Summer	Fall	Mean	Spring	Summer	Fall	Mean	Spring	Summer	Fall	Mean
Phytoplankton biomass ($10^3 mg/m^3$)	2.9	2.7	1.5	2.4	1.5	4.9	2.7	3.2	7.1	5.4	3.3	5.3
	% composition											
Cyanophyta	5.8	12.0	3.1	7.5	3.1	5.1	1.5	3.4	0.3	22.1	3.0	9.8
Chlorophyta	5.2	24.0	13.3	15.2	7.4	12.6	14.7	11.7	19.5	12.4	5.0	12.3
Chrysomonadinae	4.9	7.0	0.3	4.4	5.3	0.6	0.5	2.0	3.7	0.9	0.3	1.6
Diatomeae	35.4	8.3	74.8	36.4	41.8	41.3	76.2	55.3	50.3	44.5	83.2	57.8
Cryptomonadinae	24.0	23.5	7.9	19.0	21.2	10.1	6.4	12.3	22.3	14.1	7.8	14.7
Dinophycinae	24.5	25.0	0.4	17.5	20.9	21.4	0.4	14.9	3.6	5.6	0.3	3.4
Total phytoflagellates	53.4	55.5	8.6	40.9	47.4	32.1	7.3	29.2	29.6	20.6	8.4	20.0

and Dinophycinae) were dominant in the Eastern basin waters.

The algal biomass and its composition for the basins is summarized as seasonal averages in Table 7. Considering the basins separately, it is apparent that diatoms contributed least in the Eastern basin (36%) but showed a relatively high percentage in both the Central and Western basins. The phytoflagellates were important in all the basins but highest values were observed in the Eastern basin. The relative proportion of Chrysomonadinae (known to grow in oligotrophic environments) was high in the Eastern basin (4%) but least in the Western basin. The Cyanophyta were most significant (10%) in the Western basin.

PHYTOPLANKTON BIOMASS DISTRIBUTIONS

The horizontal distribution of phytoplankton biomass in the surface waters of Lake Erie during 1970 is shown in Fig. 6A and B. It appears that the major growth pulse had begun by April in the west end of the lake with the biomass levels remaining low in the remainder of the lake. During May the biomass increased throughout the lake. This increase continued into June in the Eastern basin but subsided across the rest of the lake. During July and August the biomass levels were generally high, but unevenly distributed throughout the lake.

There was a major bloom in the Western basin at the end of August and it seems that this was followed by an overall period of growth so that the biomass was ranging between 5 and 15 g m^{-3} throughout the lake by the end of September. From this point on, the biomass level began to diminish so that the late fall condition was a fairly even, comparatively low biomass distribution.

SEASONAL VARIATIONS

The seasonal biomass fluctuations for the entire lake (Fig. 7) indicate that high biomass concentrations were observed during spring and maximum values were recorded during August–September. The mean basin weighted and unweighted biomass concentration was 3.4 and 10.8 g/m^3 respectively. When the lake is considered as a whole the diatoms were the dominant group 53%). The importance of phytoflagellates (29%) and the low proportion of blue-greens is also apparent. The seasonal variations of chlorophyll *a*, primary production, and biomass by group composition for the three basins is shown in Fig. 8. Primary production and chlorophyll *a* generally showed seasonal patterns similar to biomass but chlorophyll indicated some discrepancies.

Two biomass maxima were observed in the Western basin; the first one in early spring (13 g/m^3) and the second one in summer (8.2 g/m^3). The spring plankton was composed of diatoms (April), phytoflagellates (May–July), and greens (May–June). However, the percentage contribution of diatoms decreased during summer due to the relative increase of greens and blue-greens. The fall phytoplankton was composed mainly of diatoms (Fig. 8).

The biomass in the Central basin started to increase from the end of June and culminated in a later summer maximum (6 g/m^3). The spring phytoplankton consisted of diatoms and phytoflagellates with the latter group being more abundant. In June, the diatoms were largely replaced by phytoflagellates. The summer plankton was

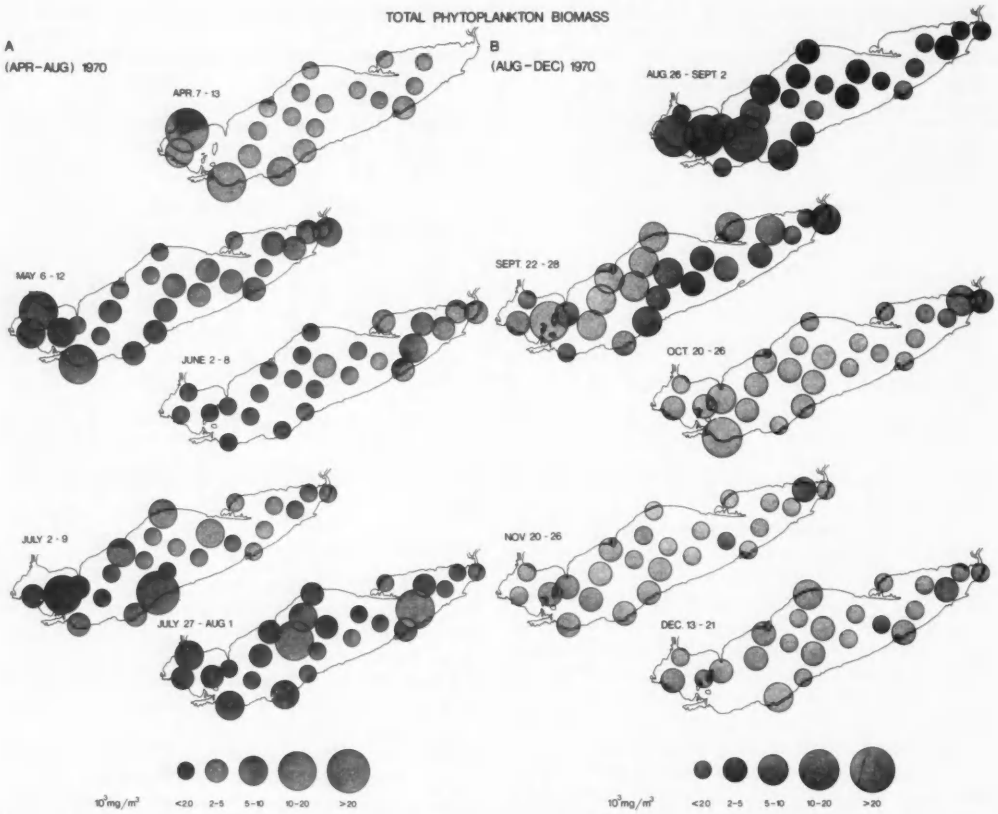

FIG. 6. A, Monthly variations of phytoplankton biomass, April–July, 1970; B, Monthly variations of phytoplankton biomass, August–December, 1970.

also dominated by phytoflagellates and diatoms whereas greens and blue-greens were less important. The early summer plankton was dominated by diatoms which decreased again by the end of July. During fall, phytoflagellates decreased and were progressively replaced by diatoms but green algae were consistently present (Fig. 8).

No major pulse was observed in the Eastern basin with three maxima of similar magnitude (4 g/m^3) being observed. The early spring samples were dominated by diatoms. In late spring (May–June), the diatoms decreased strikingly and were replaced by phytoflagellates. From the end of June onward and throughout the entire summer, diatoms were practically absent, having been replaced by phytoflagellates, greens and blue-greens. Diatoms once again dominated the phytoplankton community in the fall.

The Western and Central basins were similar in relative importance of diatoms during all seasons, whereas in the Eastern basin diatoms were practically absent over a 3-mo period from June to August. However, with respect to dinoflagellates, the Central and Eastern basins were similar and both different from the Western basin. The phytoflagellates showed a definite increasing gradient from west to east. Chlorophyta were an important component of the spring and early summer plankton in the Western basin and late summer in the Eastern basin. The blue-greens reached a distinct maximum in late summer only in the Western basin but never dominated the phytoplankton community of Eastern and Central basins.

Differences in the biomass of the basins could be related to differences in nutrient conditions. The abundance of blue-greens in the Western basin could indicate nutrient enrichment but it is not clear why the blue-greens were less important in the Central basin. The increasing relative importance of phytoflagellates from west to east,

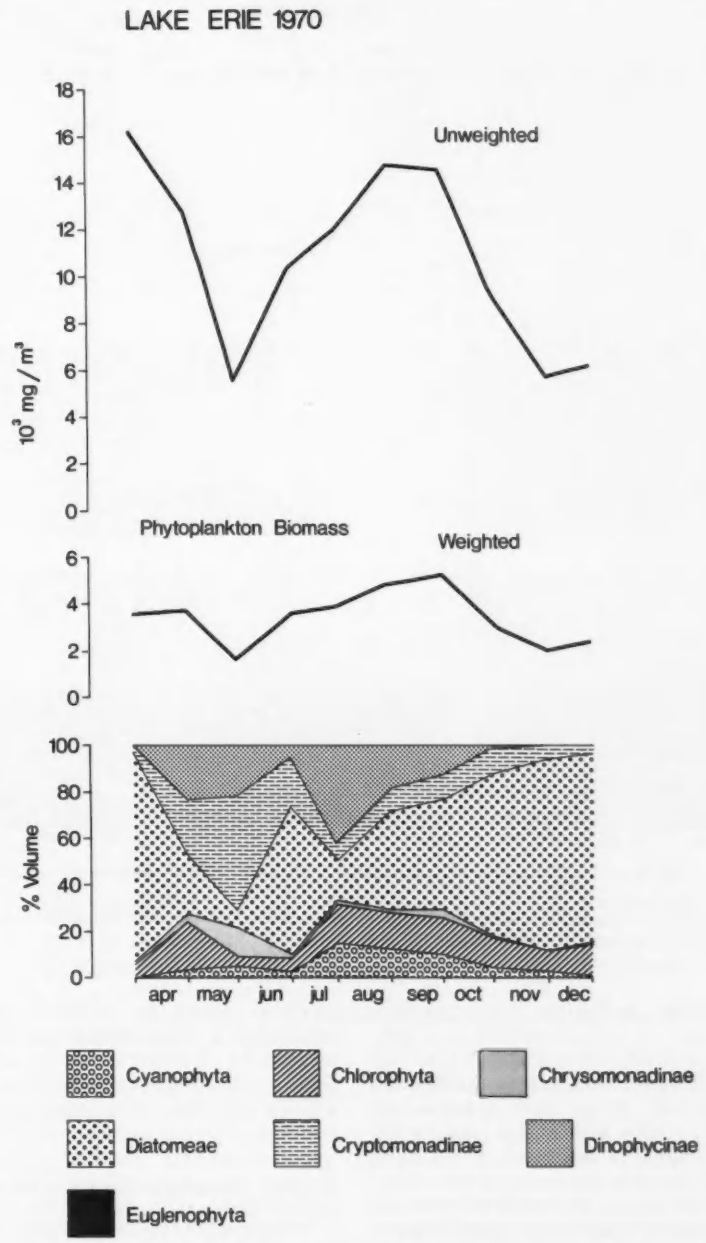

FIG. 7. Seasonal fluctuations of phytoplankton biomass and its group composition in Lake Erie, mean of all the stations sampled across the lake.

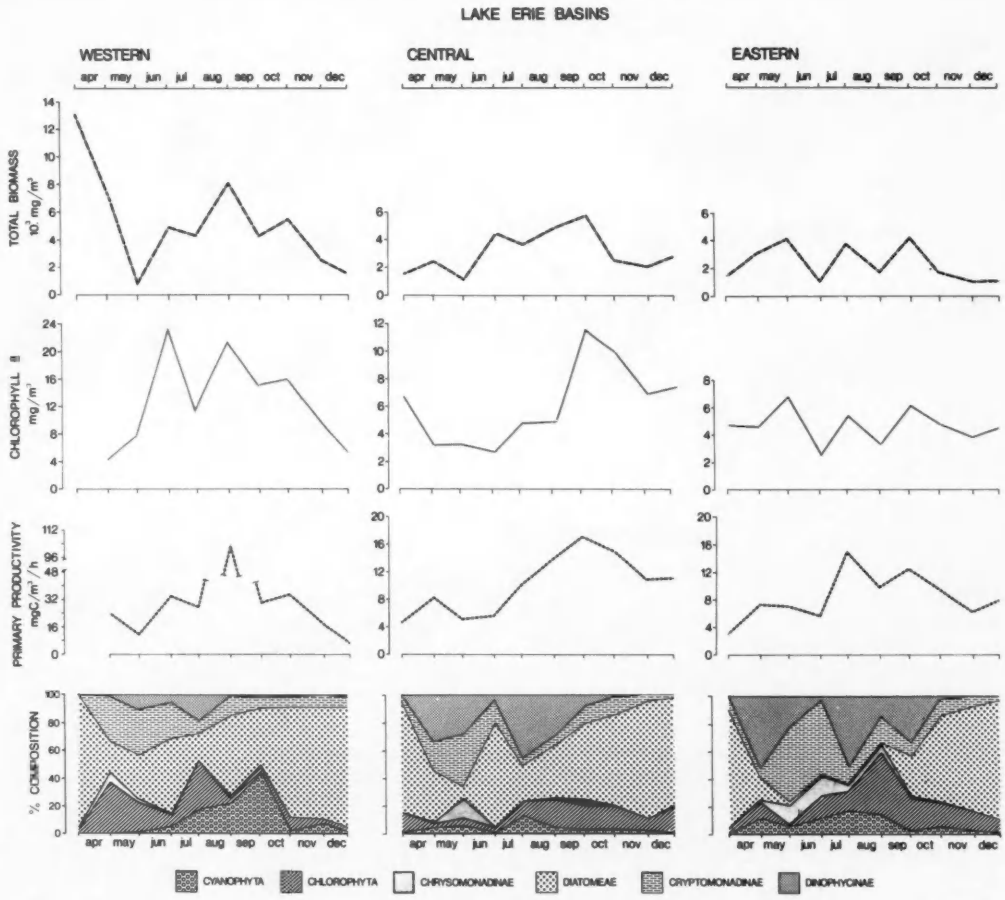

FIG. 8. Seasonal fluctuations of phytoplankton biomass, its group composition, Chlorophyll *a* (uncorrected), and primary production in Lake Erie basins, 1970.

which parallels the decreasing biomass gradient in the same direction could be related to a west–east gradient in depth, thermal stability, or nutrient concentrations. It has been suggested that phytoflagellates with their ability of locomotion could migrate into the hypolimnion, absorb soluble reactive phosphorus, and return upward again to obtain light for photosynthesis (Burns 1976). Since diatoms are heavier and nonmotile they are vulnerable to sinking. Motile phytoflagellates are better adapted to act against sinking which would be of special importance during the stratified season when the upper mixed layer is comparatively shallow. There was adequate soluble reactive silica in the epilemnion of the Eastern basin to promote diatom growth but the phytoflagellates remained dominant during the stratified period. Also, it seems that the flagellates have a wide range of adaptability to different nutrient conditions since similar species of flagellates were found at the various nutrient levels prevailing in the three basins.

INSHORE–OFFSHORE DIFFERENTIATION

Inshore–offshore differentiation similar to that observed in Lake Ontario (Munawar and Nauwereck 1971) is also apparent in the Lake Erie basins. Since western Lake Erie is shallow, well-mixed, and influenced by rivers like the Maumee and the Detroit River, it is unrealistic to categorize inshore and offshore regions in that basin.

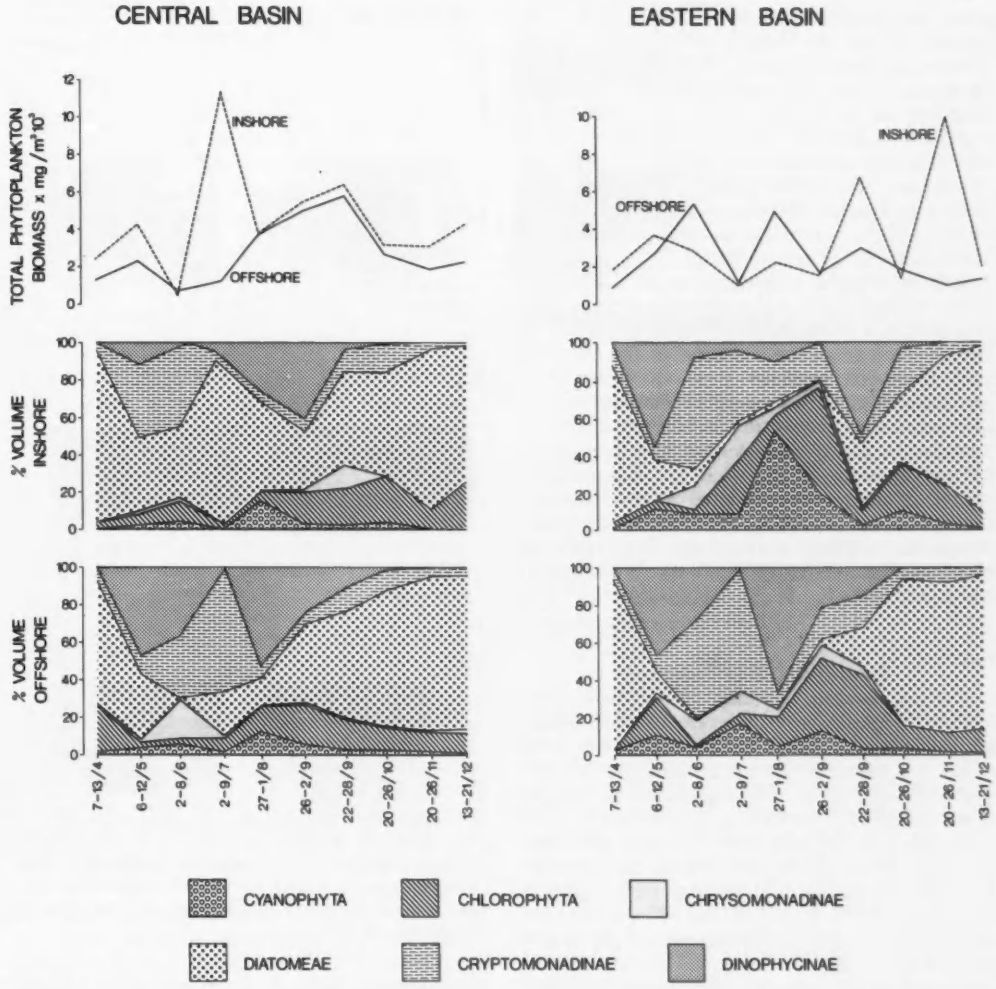

FIG. 9. Seasonal fluctuations of phytoplankton biomass and group composition in the inshore and offshore regions of Central and East basins of Lake Erie, 1970.

Therefore, the Central and Eastern basins have been chosen to evaluate differences between the inshore and offshore phytoplankton communities (Fig. 9), where the "offshore" stations are deeper than 18 m (Fig. 1). It is apparent that the mode and time of biomass pulse formation in the two regions was significantly different.

In the Central basin, three peaks were observed in the inshore region, the first one in spring and the latter during summer, with a maximum of as much as 12 g/m³ in July. The offshore region showed only a single pulse during late summer (September). It seems diatoms were the major component in the inshore region for most of the year whereas phytoflagellates were dominant in the offshore region during May–August having practically replaced the diatoms.

In the Eastern basin the inshore region was also characterized by a number of maxima with a biomass density of up to 10 g/m³. The offshore region exhibited two well-developed pulses during the spring (June) but the summer pulse did not materialze in the offshore region. Two pronounced peaks developed during fall in the inshore region which is somewhat different from the other trends observed. The Eastern basin

group composition presents an interesting comparison. In both the inshore and offshore regions, diatoms dominated during April and October–December period. The phytoflagellates were also abundant in both the regions but were more dominant in the offshore region during the summer. The inshore region though did not show the dominance of diatoms usually characteristic of inshore or shallow environments but instead contained a high percentage of blue-greens during summer (50% of total biomass).

Inshore-offshore differentiation in Lake Erie when compared to Lake Ontario is less pronounced on a lakewide basis. This may be due to the morphology or the weakness of the "thermal bar" formation in Lake Erie. However, there are differences in the seasonal development of phytoplankton in that the inshore region on a lakewide basis was characterized by a large number of biomass pulses while, in the offshore region, the biomass maxima occurred mainly during the summer (Fig. 8). These observations are similar to others on the Lake Ontario phytoplankton (Munawar and Nauwerk 1971; Munawar et al. 1974).

Phytoplankton Size Analysis

Cell size plays an important role in food chain mechanics, particularly in regard to zooplankton grazing and the transfer of pollutants and toxic substances through the organisms of smaller size (nannoplankton) which are abundant in the Great Lakes. No exact line can be drawn between the types of organisms (Pavoni 1963) and any method of separation between nannoplankton and netplankton will be somewhat artificial. For example, Rodhe (1958a) used 100 μm whereas Willen (1959) considered 60 μm as the delimiting size. In this study, the average longest dimension of 64 μ was used as the size criterion (Vollenweider et al. 1974) for net ($>$ 64 μm) and nannoplankton ($\leqslant$ 64 μm) separation. A few examples of phytoplankton size composition at various stations of the three basins are given in Fig. 10. It is apparent that nannoplankton comprised a major fraction of the phytoplankton biomass during the spring and fall periods. The samples shown in Fig. 10 also show that, although there are peaks of nannoplankton dominance in the summer, the phytoplankton biomass at this time of the year largely consisted of netplankton. A further breakdown of the nannoplankton into various size fractions (e.g. $<$10 μm, 10–20, 20–40 and 40–64 μm) is shown for a few stations in the three basins (Fig. 11). It is apparent that a wide variety of sizes composed the nannoplankton in the various seasons. In

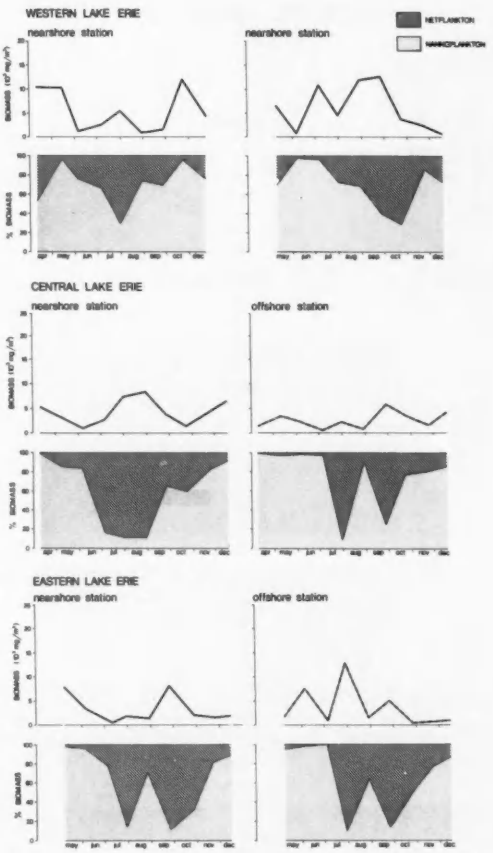

FIG. 10. Seasonal fluctuations of netplankton and nannoplankton fractions of the phytoplankton biomass at a few nearshore and offshore stations in Lake Erie basins, 1970.

general, nannoplankton maxima were composed of 20–40 μm and 40–64 μm size fractions. Fractionation experiments demonstrated that nannoplankton were most active photosynthetically. Furthermore, $<$10-μ fraction contributed significantly in the Eastern and Central basins (Munawar and Burns 1976).

Phytoplankton Biomass Comparison

As a final section in this study, a comparison is made of the maximum biomass concentrations observed in Lake Erie with those observed in the other Great Lakes environments and European lakes (Fig. 12). The Western basin of Lake Erie showed a maximum value of 13 g/m^3 which is exceeded only by that of Saginaw Bay (17.3 g/m^3). In contrast in Lake Superior, White

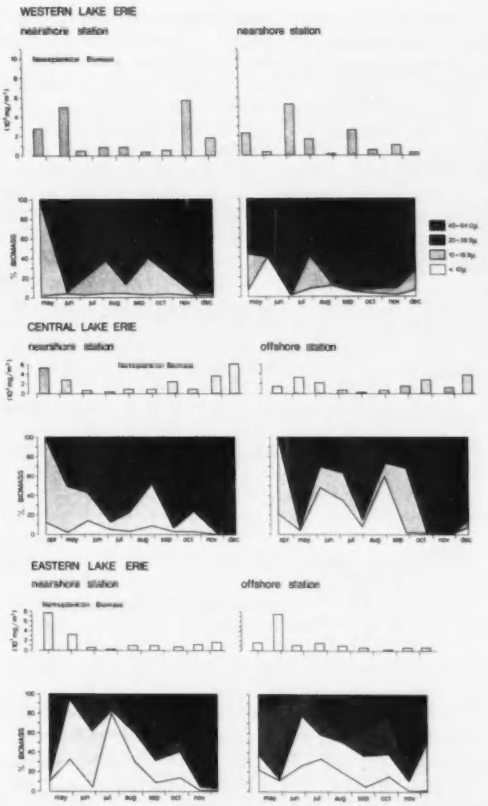

FIG. 11. Seasonal fluctuations of nannoplankton biomass and its various size fractions at a few nearshore and offshore stations in Lake Erie basins, 1970.

Fish Bay station showed a maximum concentration of 0.38 g/m³ whereas an offshore station indicated 0.19 g/m³ (Table 8). For interest, the Western basin vaiue (13.2 g/m³) is similar to the eutrophic Lake Erken in Sweden (Rodhe 1958b). However, the offshore Eastern and Central basin stations are apparently similar to Lake Constance (Elster and Motsch 1966).

Vollenweider (1968) has suggested that maximum phytoplankton biomass concentration might serve as an indicator of the trophic nature of an environment. He has classified the European lakes as follows:

Ultraoligotrophic	< 1 g/m³
Mesotrophic	3 to 5 g/m³
Highly eutrophic	> 10 g/m³

According to the above scheme Lake Erie Western basin would be classified as highly eutrophic whereas the Eastern basin would fall under the mesotrophic category. The Central basin appears

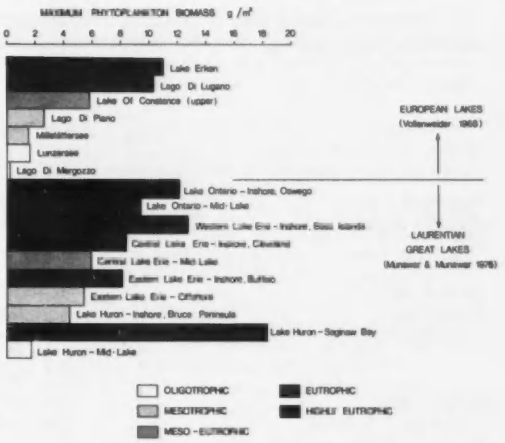

FIG. 12. Maximum phytoplankton biomass concentration observed at Lake Erie stations compared to other Great Lakes stations and European lakes.

between the mesotrophic and highly eutrophic category. This is in agreement with the classification of Vollenweider et al. (1974) based on primary production rate.

Summary and Conclusions

The past data on lakewide phytoplankton biomass and its species composition has been lacking. "Less common" species were seldom counted previously, but in this study both the "common" and "less common" species have been enumerated and the collective importance of the "less common" or rare species has been demonstrated. About 125–150 species have been identified in each basin with the highest number of species belonging to Chlorophyta. The phytoflagellates showed a 12–18% contribution to the species number in the three basins.

A comparison of the species composition amongst the three basins showed that the phytoflagellates like *Rhodomonas minuta* and *Cryptomonas erosa* were perennial in all the basins. The species *Aphanizomenon flos-aquae*, *Anabaena spiroides* var. *contracta*, *Coscinodiscus rothii*, *Actinastrum hantzschii*, and *Ceratium hirundinella* were typical for the Western basin. *Ochromonas* spp. and *Peridinium aciculiferum* were common in the Central basin and the Eastern basin was characterized by *Gymnodinium helveticum* and *G. uberrimum*.

Considering the basins separately, it was found that on an average the diatoms contributed least in the Eastern basin (36%) to the total phytoplankton biomass but showed a relatively high percentage in both the Central (55%) and West-

TABLE 8. Comparison of the range of biomass levels observed in various Great Lakes environments.

	No. stations	Date	Phytoplankton biomass ($10^3 mg/m^3$) Range	Reference
		Lake Ontario		
Nearshore region	7	13 surveys	0.8–7.4	Munawar and Nauwerck (1971)
Offshore region	20	Jan.–Dec. 1970	0.6–9.0	
		Lake Erie		
Eastern basin	7	10 Surveys	1.0–4.2	
Central basin	12	Apr.–Dec. 1970	0.6–6.0	Munawar and Munawar (1976)
Western basin	5		0.8–13.2	
		Lake Huron Stations		
Saginaw Bay	1	8 surveys	1.6–17.3	
Nearshore (off Sarnia)	1	Apr.–Dec. 1971	0.9–3.1	
Nearshore (off Bruce Peninsula)	1		0.3–4.4	Munawar and Munawar (1975)
Nearshore (Strait of Mackinac)	1		0.3–0.9	
Offshore (southern)	1		0.1–1.4	
Offshore (northern)	1		0.3–1.8	
		Lake Superior Stations		
Nearshore (Whitefish Bay)	1	6 surveys	0.05–0.58	
Nearshore (Thunder Bay, Ont.)	1	May–Nov. 1973	0.05–0.27	Munawar and Munawar (1975)
Nearshore (Duluth, Minn.)	1		0.16–0.38	
Offshore	1		0.03–0.19	

ern (58%) basins. The phytoflagellates were important in all the basins but highest values were observed in the Eastern basin (41%). The bluegreens were most significant in the Western basin in the summer (22%) but constituted only 10% of the biomass on an average. They were relatively less important in the other two basins. When the lake was considered as a whole, the diatoms (53%) and phytoflagellates (29%) were the major contributors.

The phytoflagellates showed a definite increasing gradient from west to east and appeared to have a wide range of adaptability to different nutrient conditions since they were found at the various nutrient levels prevailing in the three basins.

The horizontal distribution of phytoplankton biomass in the surface waters indicated that the major growth pulse had commenced by April in the west and with biomass concentration remaining low in the rest of the lake. The biomass increased throughout the lake in May. The July and August collections showed a generally high but nevertheless uneven distributions. Low biomass levels were again observed in late fall.

The seasonal variations in the three basins showed that two maxima were observed in the Western basin, one each in spring (13 g/m^3) and summer (8 g/m^3). In the Central basin the biomass was high in June but a peak was observed in summer (6 g/m^3). No major pulse was observed in the Eastern basin. Instead three maxima of low and similar concentrations (4 g/m^3) were recorded.

Inshore–offshore differentiation similar to that observed in Lake Ontario was also apparent in Lake Erie but was less pronounced. The inshore region indicated large numbers of biomass peaks.

The size analysis of the total phytoplankton community at a few stations indicated that the nannoplankton ($\leq 64\ \mu$) comprised a major portion of the biomass during spring and fall periods. The netplankton ($> 64\ \mu$) contributed most to the summer biomass. Furthermore, a wide variety of sizes composed the nannoplankton in various seasons.

A comparison of the present study was made with other Great Lake and European environments. Based on maximum biomass concentration the Lake Erie Western basin was classified as highly eutrophic, the Eastern basin as mesotrophic, and the Central Basin ranged between the mesotrophic and eutrophic conditions.

Acknowledgments

The study was carried out during the tenure of a postdoctoral research fellowship (1969–71) awarded to M. Munawar by the National Research Council of

Canada, under the supervision of Dr R. A. Vollenweider, Senior Scientist, CCIW. We are grateful for his guidance and continued interest in our research. We gratefully acknowledge the advice and encouragement given by Drs A. Nauwerck, C. Nalewajko, and D. Cook. We thank G. Dupuis, L. R. Culp, and P. Young for their assistance; and B. Finn and drafting staff for the preparation of diagrams. The editorial help of Dr N. Burns which improved the clarity of this manuscript is greatly appreciated. We are grateful to Prof. J. Verduin for inspiring our interest in Lake Erie during our association with him at Southern Illinois University.

BRAIDECH, T. E., P. E. GEHRING, AND C. O. KLEVENO. 1971. Biological studies of oxygen depletion and nutrient regeneration processes in the Lake Erie Central basin. Proc. Conf. Great Lakes Res. 14: 805–817.

BURKHOLDER, P. R. 1929a. Microplankton studies in Lake Erie. Bull. Buffalo Soc. Nat. Sci. 14: 73–93.

1929b. Microplankton studies in Lake Erie. N.Y. Conserv. Dep. Annu. Rep. Suppl. 18: 60–66.

1960. A survey of the microplankton of Lake Erie. U.S. Fish. Wildl. Serv. Spec. Sci. Fish Rep. 334: 123–144.

BURNS, N. M. 1976. Temperature, oxygen, and nutrient distribution patterns in Lake Erie, 1970. J. Fish. Res. Board Can. 33: 485–511.

CASPER, V. L. 1965. A phytoplankton bloom in Western Lake Erie. Proc. 8th Conf. Great Lakes Res. 8: 29–35.

CHANDLER, D. C. 1940. Limnological studies of Western Lake Erie. I. Plankton and certain physical chemical data of the Bass islands region, from September 1938 to November 1939. Ohio J. Sci. 40: 291–336.

1942. Limnological studies of Western Lake Erie. III. Phytoplankton and physical chemical data from November 1939 to November 1940. Ohio J. Sci. 42: 24–44.

1944. Limnological studies of Western Lake Erie. IV. Relation of limnological and climatic factors to phytoplankton of Western Lake Erie. Trans. Am. Microsc. Soc. 63: 203–236.

DAVIS, C. C. 1954a. A preliminary study of the plankton of the Cleveland harbour area, Ohio. II. The distribution and quantity of the phytoplankton. Ecol. Monogr. 23: 321–347.

1954b. A preliminary study of the plankton of the Cleveland harbour area, Ohio. III. The zooplankton and general ecological considerations of phytoplankton and zooplankton production. Ohio J. Sci. 54: 388–408.

1964. Evidence for the eutrophication of Lake Erie from phytoplankton records. Limnol. Oceanogr. 9: 275–283.

1965. The standing stock of phytoplankton in Lake Erie at Cleveland, Ohio, 1964. Inf. Bull. Planktol. Japan 12: 51–53.

1966. Plankton studies in the Great Lakes of the world with special reference to the St. Lawrence Great Lakes of North America. Great Lakes Res. Div. Univ. Mich. Publ. 14: 54 p.

1969. Plants in Lakes Erie and Ontario and changes of their numbers and kinds. Proc. Conf. on changes in the Biota of Lakes Erie and Ontario. Bull. Buffalo Soc. Nat. Sci. 25: 18–44.

ELSTER, H. J., AND B. MOTSCH. 1966. Untersuchungen über das Phytoplankton und die organische Urproduktion in einigen Seen des Hochschwarzwalds, im Schleinsee und Bodensee. Arch. Hydrobiol. Suppl. 28: 291–376.

GLOOSCHENKO, W. A., J. E. MOORE, M. MUNAWAR, AND R. A. VOLLENWEIDER. 1974a. Primary production in Lakes Ontario and Erie: a comparative study. J. Fish. Res. Board Can. 31: 253–263.

GLOOSCHENKO, W. A., J. E. MOORE, AND R. A. VOLLENWEIDER. 1974b. Spatial and temporal distribution of chlorophyll *a* and pheopigments in surface waters of Lake Erie. J. Fish Res. Board Can. 31: 265–274.

GOTTSCHALL, R. Y. 1930. Preliminary report on the phytoplankton and pollution in Presque Isle Bay, Lake Erie. Proc. Pa. Acad. Sci. 4: 69–70.

GOTTSCHALL, R. Y., AND O. E. JENNINGS. 1933. Limnological studies at Erie, Pa. Trans. Am. Microsc. Soc. 52: 181–191.

HOHN, M. H. 1969. Qualitative and quantitative analyses of plankton diatoms, Bass Island area, Lake Erie, 1938–1965, including synoptic surveys of 1960–1963. Ohio State Univ., Bull. Ohio Biol. Surv. New Ser. 3: 211.

KELLICOT, D. S. 1878. Notes on microscopic life in the Buffalo water supply. Am. J. Microsc. Pop. Sci. 3: 250–252.

LUND, J. W. G., C. KIPLING, AND E. D. LECREN. 1958. The inverted microscope method of estimating algal numbers and the statistical basis of estimations by counting. Hydrobiologia 11: 43 p.

MILLS, H. 1882. Microscopic organisms in the Buffalo water supply and in Niagara river. Proc. Am. Soc. Microsc. 5: 165–175.

MUNAWAR, M., AND N. M. BURNS. 1976. Relationships of phytoplankton biomass with soluble nutrients, primary production, and chlorophyll *a* in Lake Erie, 1970. J. Fish. Res. Board Can. 33: 601–611.

MUNAWAR, M., AND I. F. MUNAWAR. 1975. The abundance and significance of phytoflagellates and nannoplankton in the St. Lawrence Great Lakes, I. Phytoflagellates. Verh. Int. Ver. Limnol. 19: 705–723.

MUNAWAR, M., AND A. NAUWERCK. 1971. The composition and horizontal distribution of phytoplankton in Lake Ontario during the year 1970. Proc. Conf. Great lakes Res. 14: 69–78.

MUNAWAR, M., STADELMANN, P., AND I. F. MUNAWAR. 1974. Phytoplankton biomass, its species composition and primary production at a nearshore and mid-lake station of Lake Ontario during IFYGL. Proc. Conf. Great Lakes Res. 17: 629–652.

NAUWERCK, A. 1963. The relation between zooplankton and phytoplankton in Lake Erken. Symb. Bot. Ups. 17: 163. (In German)

OLIVE, J. H., D. M. BENTON, AND J. KISHLER. 1969. Distribution of C-14 in products of photosynthesis and its relationship to phytoplankton composition and rate of photosynthesis. Ecology 50: 380–386.

PAVONI, M. 1963. Die bedeutung des Nannoplanktons im Vergleich zum Netplankton. Schweiz. Z. Hydrol. 25: 219–341.

RODHE, W. 1958a. Primarproduktion und Seetypen. Verh. Int. Ver. Limnol. 13: 121–141.

1958b. The primary production in lakes: some

results and restrictions of the C-14 method. Rapp. Proc. Verb. Cons. Perm. Int. Explor. Mer 14: 122–128.

TAFT, C. E., AND C. W. TAFT. 1971. The algae of Western Lake Erie. Ohio State Univ., Bull. Ohio Biol. Surv. New Ser. 4: 189.

TIFFANY, L. H. 1934. The plankton algae of the West end of Lake Erie. Contrib. F. T. Stone Lab. 6: 112 p.

1937. The filamentous algae of the West end of Lake Erie. Contrib. No. 6, Franz Theodor Laboratory, Ohio State University Press, Columbus, Ohio. 112 p.

UTERMÖHL, H. 1958. Zur Vervolkommung der quantitativen phytoplankton. Methodik. Mitt. Int. Ver. Limnol. 9: 1–38.

VERDUIN, J. 1951. Comparison of spring diatom crops of Western Lake Erie in 1949 and 1950. Ecology 32: 662–668.

1952. Photosynthesis and growth rates of two diatom communities in Western Lake Erie. Ecology 33: 163–168.

1960. Phytoplankton communities of Western Lake Erie and the CO_2 and O_2 changes associated with them. Limnol. Oceanogr. 5: 372–380.

1964. Changes in Western Lake Erie during the period 1948–1962. Verh. Int. Ver. Limnol. 15: 639–644.

VOLLENWEIDER, R. A. 1968. Scientific fundamentals of the eutrophication of lakes and flowing waters, with particular references to nitrogen and phosphorus as factors in eutrophication. Organ. Econ. Coop. Dev. Rep., Paris.

1969. A manual on methods for measuring primary production in aquatic environments. IBP Handb. No. 12. Blackwell Scientific Publications, Oxford. 213 p.

VOLLENWEIDER, R. A., M. MUNAWAR, AND P. STADELMANN. 1974. A comparative review of phytoplankton and primary production in the Laurentian Great Lakes. J. Fish. Res. Board Can. 31: 739–762.

WILLEN, T. 1959. The phytoplankton of Gorwälm, a bay of Lake Malaren. Oikos 10: 241–274.

WRIGHT, S., L. H. TIFFANY, AND W. M. TIDD. 1955. Limnological survey of Western Lake Erie. U.S. Fish. Wildl. Serv. Spec. Sci. Fish Rep. 139: 1–341.

WUJECK, D. E. 1967. Some plankton diatoms from the Detroit River and the Western end of Lake Erie adjacent to Detroit River. Ohio J. Sci. 67: 32–35.

Relationships of Phytoplankton Biomass with Soluble Nutrients, Primary Production, and Chlorophyll *a* in Lake Erie, 1970

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MUNAWAR, M., AND N. M. BURNS. 1976. Relationships of phytoplankton biomass with soluble nutrients, primary production, and chlorophyll *a* in Lake Erie, 1970. J. Fish. Res. Board Can. 33: 601–611.

Comparison of the annual average distribution patterns of phytoplankton biomass, chlorophyll *a*, primary production, soluble reactive phosphorus, nitrate + nitrite, and ammonia concentrations revealed that these six variables had very similar distributions in Lake Erie during 1970. However, statistical analysis of the data only revealed a few consistent relationships between these variables. The phytoplankton biomass was correlated with chlorophyll *a* only in the summer and fall as was primary production with chlorophyll *a* and biomass. There was no correlation between these three variables during the spring. Also, there was no consistent relationship between biomass and soluble nutrients. The primary production and activity coefficient (mg $C_{assimilated}$ per milligram phytoplankton biomass per day) were found to be unrelated to temperature. The statistical procedure of factor analysis showed that in the spring, primary production correlated with the phosphorus and nitrogen soluble nutrients only, whereas during summer, primary production correlated with biomass, chlorophyll *a*, the major plankton groups (Cyanophyta, Chlorophyta, Chrysomonadinae, and Diatomeae), and the phosphorus nutrients. In the fall, production was positively correlated with phytoplankton biomass and with the Chlorophyta in particular. The use of chlorophyll *a* and temperature as variables in the equation to estimate phytoplankton growth in Lake Erie was found to be questionable.

MUNAWAR, M., AND N. M. BURNS. 1976. Relationships of phytoplankton biomass with soluble nutrients, primary production, and chlorophyll *a* in Lake Erie, 1970. J. Fish. Res. Board Can. 33: 601–611.

Une comparaison de la distribution moyenne annuelle de la biomasse de phytoplancton, de la chlorophylle *a*, de la production primaire, du phosphore réactif soluble, des teneurs en nitrates + nitrites et en ammoniaque démontre que ces six variables ont des distributions très semblables dans le lac Erié en 1970. La corrélation positive anticipée de ces variables entre elles n'a pu être démontrée par analyse statistique des données, qui ne révèlent que quelques relations uniformes entre ces variables. La biomasse de phytoplancton est liée à la chlorophylle *a* seulement en été et en automne. Il en est de même de la relation entre la production primaire et la chlorophylle *a* et la biomasse. Il n'y a pas de corrélation entre ces trois variables au printemps. De plus, il n'y a pas de relation uniforme entre la biomasse et les substances nutritives solubles. La production primaire et le coefficient d'activité (mg $C_{assimilé}$ par milligramme de biomasse de phytoplancton par jour) s'avèrent indépendants de la température. L'analyse statistique démontre qu'au printemps, la production primaire n'est liée qu'aux substances nutritives solubles phosphorées et azotées, alors qu'en été, la production primaire est liée à la biomasse, à la chlorophylle *a*, aux groupes majeurs de plancton (cyanophytes, chlorophytes, chrysomonadinées et diatomées) et aux substances nutritives phosphorées. En automne, la production est en corrélation positive avec la biomasse de phytoplancton et avec les chlorophytes en particulier. L'emploi de la chlorophylle *a* et de la température dans l'équation servant à estimer la croissance du phytoplancton dans le lac Erié est discutable.

Accepted November 18, 1975

Accepté le 18 novembre 1975

DATA on the phytoplankton biomass down to the species level have recently become available for all of Lake Erie (Munawar and Munawar 1976). Concentrations of soluble nutrients in the phytoplankton samples during each of the 10 sampling surveys in 1970 were also measured (Gächter et al. 1974). In addition, data on the primary production (Glooschenko et al. 1974a) and the temperature and chlorophyll *a* concentrations within the same samples (Glooschenko et al. 1974b) were measured. This data assemblage now makes it possible to examine the relationships prevailing in Lake Erie between phytoplankton biomass of various taxonomic groups and species with soluble nutrients, temperature,

Printed in Canada (J4134)
Imprimé au Canada (J4134)

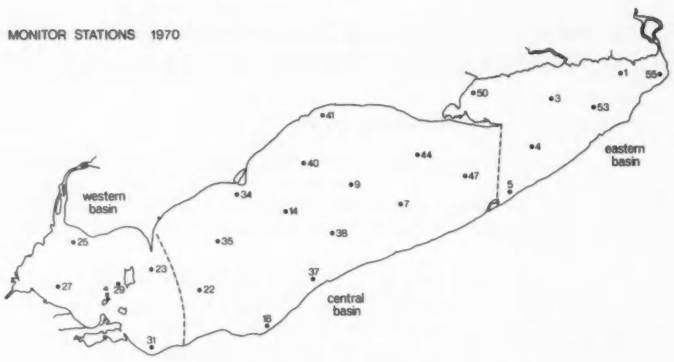

FIG. 1. Lake Erie showing the biological sampling stations.

primary production, and chlorophyll *a*. The results of this investigation are reported in this paper.

Methods

The stations that were sampled for phytoplankton during the 10 surveys of Lake Erie from April to December 1970 are shown in Fig. 1. Water was drawn from the lake at the 1.0- and 5.0-m depths and subsequently mixed. This sample then provided the subsamples for the analyses of the phytoplankton biomass (Munawar and Munawar 1976), the soluble nutrients (Gächter et al. 1974) primary production (Glooschenko et al. 1974a), and total chlorophyll *a* (Glooschenko et al. 1974b). The primary production experiments were conducted in a specially constructed incubator on board the survey ship with the samples kept at the temperature of the surface lake water which was continuously recorded. The activity coefficient (mg $C_{assimilated}$ per milligram biomass per day) was calculated by dividing the measured primary production rate by the biomass concentration, assuming an average light period of 10 h/day.

Experiments that tested the primary production of different size fractions were done in July 1973 using surface samples which were subdivided among 15 light bottles and 5 dark bottles. Each bottle was inoculated with approximately 2 μC. of $Na_2\ ^{14}CO_3$ and the light bottles were incubated for 2–4 h in a light box (40.1×10^{11} ergs/m^2/h) similar to that described by Stadelmann and Moore (1974). After incubation, the bottles were grouped into five sets consisting of three light bottles and one dark bottle each. The first set was filtered directly through 0.45-μ pore size membrane filters. Sets 2, 3, 4, and 5 were washed through a 64-μ, 44-μ, 20-μ, and 10-μ nitex screen, respectively, and subsequently filtered through 0.45-μ pore size membrane filters. The filters were analyzed for activity determinations by the liquid scintillation counting procedure (Lind and Campbell 1969).

Mean annual distribution maps for the phytoplankton-related variables were drawn (Fig. 2) by using the annual average values observed at the 25 sampling stations. The annual average nutrient concentration maps (Fig. 3) were drawn using the average epilimnion concentration data observed at the 56 stations including the 25 phytoplankton stations (Burns 1976). A comparison of the distribution maps of Gächter et al. (1974) with 25 sampling stations and Burns (1976) with 56 sampling stations, showed only minor differences. Since the data of Gächter et al. (1974) was obtained from the samples used for phytoplankton biomass, chlorophyll *a*, and primary production measurements, these nutrient data were used in the statistical analysis of the variables.

An outline of the methods and programs used in the factor analysis can be found in Jaquet et al. (1975) and Cameron (1967). The 17 original variables were log-transformed to normalize the distributions. The computed correlation coefficients were tested for significance at 0.05 probability level. Preliminary runs indicated that the correlations between variables changed significantly with the time of the year, sometimes making the analysis of the entire data set meaningless (Table 1). The data were therefore split into three seasonal sets: spring (April–June), summer (July–September), and fall (October–December) and factor analysis was carried out on the three sets separately. Partial correlation calculations were also done on the biomass-soluble nutrient data from the 13 Central basin stations for each of the 10 surveys. These calculations were limited to biomass and the five other variables shown in Table 2.

Results

Mean annual distributions of phytoplankton biomass, chlorophyll *a*, and primary production are shown in Fig. 2, and the same type of distributions for soluble reactive phosphorus, nitrate plus nitrite, and ammonia are shown in Fig. 3. The similarities between these averaged distributions are striking and it appears that all these variables were directly correlated with each other.

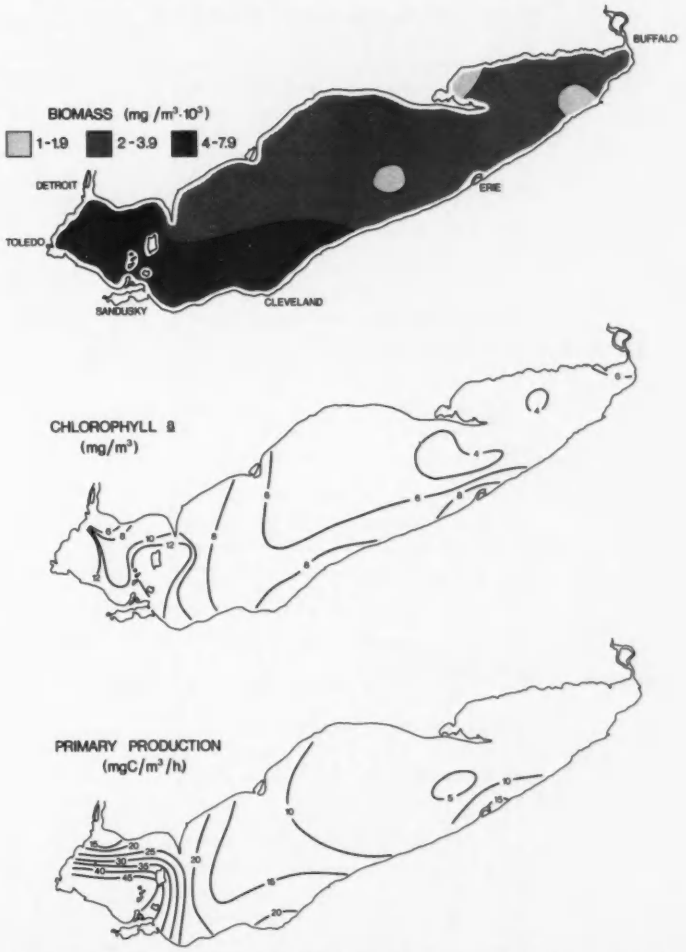

FIG. 2. Distribution maps showing the average concentrations of phytoplankton biomass, total chlorophyll *a*, and primary production observed during 10 surveys of Lake Erie during 1970.

However, statistical comparison of the values measured for each of these variables on a sample by sample basis revealed that the relationships were not as simple as would appear from the comparison of the annual average values. The results of the detailed analyses are presented below:

PHYTOPLANKTON–NUTRIENT RELATIONSHIPS

Data in Table 1 show that, when the entire data set (A = annual) was subjected to correlation analysis, there was no significant correlation between phytoplankton biomass and soluble nutrients. However, there was a positive correlation between diatoms and soluble nutrients and an inverse correlation existing between the chrysomonadinae and soluble nutrients. These differing correlations probably caused the lack of correlation between biomass and nutrients when the data were analyzed on an annual basis.

It was thought that the lack of correlation between phytoplankton biomass and soluble nutrients could have resulted from using a data set based on the three different basins of the lake and hence analyses were done on the data from the Central basin only for each survey. The results are shown in Table 2 and demonstrate quite

TABLE 1. Simple correlation coefficients for both annual and seasonal data sets. Sp, Sring; Sr, Summer; F, Fall; A, Annual (10 cruises). (0, no correlation; +, positive correlation; −, negative correlation [at 0.05 significance level].)

		Total biomass				CHLO				PRIP				TEMP				CYAN				CLOR				CHRY				CRYP				DINO				DIAT			
		Sp	Sr	F	A	Sp	Sr	F	A	Sp	Sr	F	A	Sp	Sr	F	A	Sp	Sr	F	A	Sp	Sr	F	A	Sp	Sr	F	A	Sp	Sr	F	A	Sp	Sr	F	A	Sp	Sr	F	A
Chlorophyll	(CHLO)	0	+	+	+	///	///	///	///	0	+	+	+	−	0	0	0	−	+	0	0	0	+	0	+	−	+	0	0	0	0	0	0	−	0	0	−	0	+	+	+
Primary production	(PRIP)	0	+	+	+	0	+	+	+	///	///	///	///	0	+	0	+	0	+	0	+	0	+	+	+	0	+	0	0	0	0	0	0	0	0	0	0	+	+	0	+
Temperature	(TEMP)	−	0	0	0	−	0	0	0	0	+	0	+	///	///	///	///	+	+	+	+	−	+	0	+	+	0	+	−	+	−	+	+	+	+	0	+	−	0	0	−
Particulate phosphorus	(PARP)	0	+	0	0	0	+	0	+	+	+	+	+	+	0	−	−	0	0	0	0	0	+	0	0	0	0	0	−	0	0	0	0	0	0	0	0	+	0	+
Soluble reactive phosphorus	(SRP)	0	+	0	0	0	+	0	+	+	+	0	+	0	0	−	−	0	0	0	0	0	0	0	0	−	0	0	−	0	0	0	−	0	0	−	0	0	+	0	+
Total filtered phosphorus	(TFP)	0	0	0	0	0	0	0	+	+	0	0	+	0	0	−	−	0	0	−	0	0	0	0	0	0	−	0	−	0	0	−	0	0	0	−	0	+	0	+
Nitrite and nitrate	(NINT)	+	0	0	0	+	0	0	+	+	0	0	+	−	0	−	−	0	0	−	−	+	0	0	0	−	−	0	−	0	0	−	−	−	0	0	−	+	+	0	+
Ammonia	(NH)	0	0	0	0	+	0	0	0	+	0	0	0	0	0	0	0	0	0	0	0	0	0	0	−	−	−	0	−	0	0	−	−	0	0	0	0	0	0	0	0
Reactive silica	(RSIO)	−	0	0	0	0	0	0	0	0	0	−	+	+	+	−	+	0	0	−	0	0	0	−	+	0	0	0	0	0	0	−	0	0	+	0	0	0	0	0	0
Dissolved organic phosphorus	(DOP)	0	0	0	0	0	0	0	0	+	0	0	0	0	0	−	−	0	0	−	0	0	0	0	0	0	−	0	−	0	0	0	0	0	0	0	0	0	0	0	+
Cyanophyta	(CYAN)	0	+	0	+	−	+	0	0	0	+	0	+	+	+	+	+	///	///	///	///	0	+	0	+	+	+	+	+	+	0	+	+	+	+	0	0	0	0	0	0
Chlorophyta	(CLOR)	0	+	+	+	0	+	0	+	0	+	+	+	−	+	0	+	0	+	0	0	///	///	///	///	0	+	0	0	0	0	0	+	−	+	0	0	+	+	0	+
Chrysomonadinae	(CHRY)	0	+	0	0	−	+	0	0	0	+	0	0	+	0	+	+	+	+	+	+	0	+	0	0	///	///	///	///	+	0	+	+	+	0	0	+	−	0	0	0
Cryptomonadinae	(CRYP)	0	+	0	+	0	0	0	0	0	0	0	0	+	−	+	+	+	0	+	+	0	0	0	+	+	0	+	+	///	///	///	///	+	0	0	+	0	+	0	0
Dinophycinae	(DINO)	0	+	0	+	−	0	0	−	0	0	0	0	+	+	0	0	+	+	0	+	−	+	0	0	+	0	0	+	+	0	0	+	///	///	///	///	0	0	0	0
Diatomeae	(DIAT)	+	+	+	+	0	+	+	+	+	+	0	+	−	0	−	−	0	0	0	0	+	+	0	+	−	0	0	0	0	+	0	0	0	0	0	0	///	///	///	///

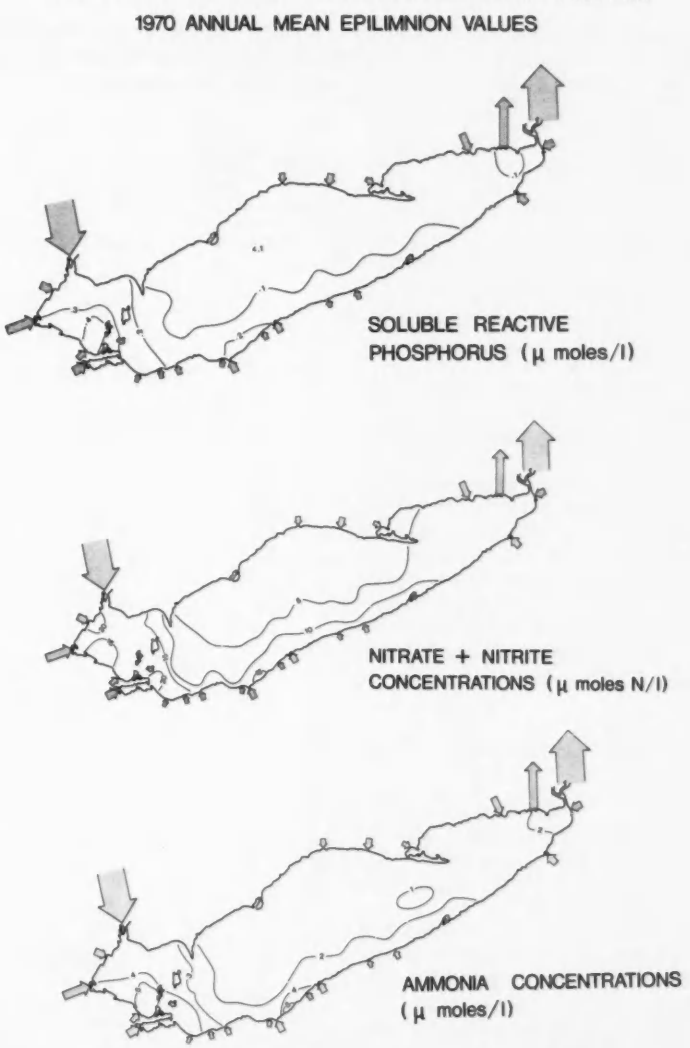

FIG. 3. Average epilimnion concentrations of soluble reactive phosphorus, nitrate plus nitrite, and ammonia observed during 10 surveys of Lake Erie during 1970.

clearly that there was no predictable relationship, in the surface waters of the basin, between the phytoplankton biomass and the soluble nutrients. Table 2 shows an almost equal number of positive and negative correlations between the biomass and soluble nutrients, with only 10% of the correlations being significant at the .05 confidence level.

PHYTOPLANKTON–TEMPERATURE RELATIONSHIPS

There was no general relationship between phytoplankton biomass and temperature (Table 1). However, in this case, the cause for the lack of the relationship can be seen in the phytoplankton group–temperature relationships. The annual correlation reveals that the strongly dominant diatoms and chrysomonadinae are significantly negatively correlated with temperature while the four less dominant groups are positively correlated, resulting in no significant biomass–temperature relationship.

Primary production is considered to increase

TABLE 2. Partial correlation coefficients[a] of biomass vs. soluble nutrients for the Central Basin, 1970.

14 stations	Apr. 7	May 5	June 2	July 3	Aug. 4	Aug. 24	Sept. 23	Oct. 21	Nov. 18	Dec. 14
Temp	0.06	−0.69	0.16	−0.56	−0.87[b]	0.66	−0.16	−0.30	−0.37	−0.43
SRP	−0.04	−0.04	−0.18	0.28	0.50	−0.07	−0.10	0.74[b]	0.39	0.34
$NO_3 + NO_2$	0.16	0.84[b]	0.39	0.44	−0.74[b]	−0.70	0.20	−0.74	−0.22	0.08
NH_4	0.11	−0.72	−0.42	−0.23	0.91[b]	0.15	0.08	0.20	−0.38	0.11
$RSiO_2$	0.21	0.09	−0.20	0.41	0.34	0.84	−0.67	−0.84	−0.67	0.06

[a]Each value represents the correlation of biomass with the variable on that particular line, with the effect of the correlation of biomass with the other variables removed.
[b]Significant at the .05 confidence level ($r = 0.71$).

with temperature but there was no positive correlation existing between these two variables during the spring and fall seasons (Table 1). Further, high values for the activity coefficient were observed in the Central basin in April (Fig. 4, Table 3) when the mean water temperature was 1.7 C and in the Western basin in November (water temperature 4.1 C) These incidences of high activity coefficient resulted in relatively high productivity at low temperatures and removed the possibility of a direct correlation between productivity and temperature. This possibility was further removed by the observation of low activity coefficients at high temperature (Fig. 4, Table 3).

PHYTOPLANKTON–PRIMARY PRODUCTION RELATIONSHIPS

The factor analysis for spring (Table 5) shows that primary production in the spring was not correlated with biomass but was apparently dependent on both phosphorus and soluble inorganic nitrogen availability (factor 1).

In the summer the particulate phosphorus and soluble reactive phosphorus are seen to have also been correlated with primary production. The primary production was directly covariant with biomass and with four of the major phytoplankton groups (Tables 6, factor 1).

It is probable that the smaller species of these groups were active since the results of the size fractionation experiment, Table 4, show that the nannoplankton were the most photosynthetically active. This is further demonstrated by the data in Table 3, where the productivity is seen to be associated with nannoplankton abundance in most cases.

In the fall the primary production remained directly covariant with the total biomass (Table 7, factor 2) but was evidently unrelated to the nitrogen and phosphorus nutrient supply. The Chlorophyta were correlated with the primary production and were evidently significantly photosynthetically active.

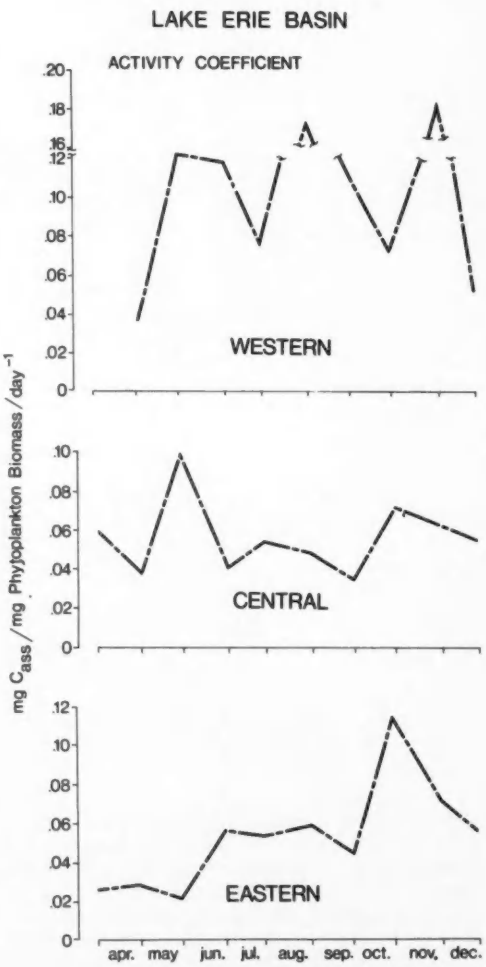

FIG. 4. Basin average values of the activity coefficient measured during primary production experiments conducted in a shipboard incubator during 1970 in Lake Erie.

TABLE 3. Examples of the temperature and species composition in samples used for experiments to determine the activity coefficient of phytoplankton biomass in Lake Erie.

Date	Basin	Temp (C)	Species	Percent biomass	Activity coefficient per day
Nov. 1970	Western	1.4	*Stephanodiscus niagarae*[a]	18.5	
			Stephanodiscus tenuis[a]	20.0	
			Coscinodiscus rothii[a]	14.0	0.19
			Melosira islandica	10.5	
			Cryptomonas erosa[a]	6.5	
July 1970	Western	23.8	*Coscinodiscus rothii*[a]	23.0	
			Cryptomonas erosa[a]	22.0	
			Stephanodiscus niagarae[a]	9.5	0.07
			Rhodononas minuta[a]	8.5	
			Aphanizomenon flos-aquae	9.0	
Apr. 1970	Central	1.7	*Stephanodiscus tenuis*[a]	34.0	
			Rhodomonas minuta[a]	8.0	
			Chlorella sp.[a]	20.5	0.06
			Stephanodiscus hantzschii[a] var. *pusilla*	12.5	
			Chlamydomonas globosa[a]	7.0	
			Melosira binderana	5.0	
Sept. 1970	Central	20.6	*Fragilaria capucina*	37.0	
			Stephanodiscus niagarae[a]	11.0	0.03
			Ceratium hirundinella	8.5	
			Fragilaria crotonensis	7.5	
Oct. 1970	Eastern	14.7	*Cryptomonas erosa*[a]	15.0	
			Fragilaria crotonensis	11.5	
			Fragilaria capucina	10.0	
			Pediastrum simplex	8.0	0.11
			Staurastrum paradoxum	8.5	
			Rhodomonas minuta[a]	9.0	
			Stephanodiscus niagarae[a]	10.0	
Sept. 1970	Eastern	19.4	*Ceratium hirundinella*	22.0	
			Fragilaria crotonensis	18.0	
			Rhodomonas minuta[a]	18.5	0.04
			Cosmarium sp.[a]	10.5	
			Staurastrum paradoxum	7.0	

[a]Indicates nannoplankton.

PHYTOPLANKTON BIOMASS–CHLOROPHYLL *a* RELATIONSHIPS

In the spring chlorophyll *a* was not correlated with total biomass (Table 1). The reason for this is probably the abundance of diatoms and cryptomonads (Munawar and Munawar 1976) which were not correlated with chlorophyll *a* (Table 1).

In the summer chlorophyll *a* was directly correlated with total biomass and with four of the major phytoplankton groups (Cyanophyta, Chlorophyta, Chrysomonadinae and diatomeae. Table 6, factor 1).

The only group correlated with chlorophyll *a* in the fall was diatoms (Table 1) and since this group was strongly dominant at the time, biomass was also correlated with chlorophyll *a* (Table 7, factor 3).

It is difficult to understand why there should be a relationship between chlorophyll *a* and biomass in the summer and fall but not in the spring. A similar pattern of noncorrelation in the spring with correlation in summer and fall, has been observed in the Central basin between chlorophyll *a* and primary production. (Glooschenko 1975).

Discussion

Comparison of the six distribution maps (Fig. 2 and 3) suggests that biomass was positively correlated with the other five variables. The results of statistical analysis show that, in the summer and fall seasons, biomass was correlated with chlorophyll *a* and primary production but that this was not the case in the spring. Out of the

TABLE 4. Results of size fractionation experiments to determine the percentage contribution of various size fractions of phytoplankton to the total primary production.

	Netplankton >64 μ	Nannoplankton <64 μ	<10 μ	10–<20 μ	20–<44 μ	44–<64 μ
Eastern basin Near Stn. #4	30.2	69.8	41.9	6.2	21.7	0.0
Central basin Near Stn. #40	23.7	76.3	37.4	3.8	27.5	7.6
Western basin Near Stn. #29	2.3	97.7	19.5	13.2	62.1	2.9

TABLE 5. Factor analysis of 17 variables measured during the spring of 1970 in Lake Erie.

		Factor					
		1	2	3	4	5	
Sum of squares		3.572	3.427	2.581	1.881	1.771	
Variable		Factor Loadings					Communality
Chlorophyll	(CHLO)	0.250	−0.664	−0.262	0.026	−0.130	0.589
Primary production	(PRIP)	0.675	0.176	−0.099	0.166	0.245	0.583
Temperature	(TEMP)	0.209	0.525	0.663	0.164	−0.320	0.889
Particulate phosphorus	(PARP)	0.800	0.066	0.050	0.156	−0.333	0.782
Total filtered phosphorus	(TFP)	0.549	−0.069	0.005	0.811	−0.026	0.964
Soluble reactive phosphorus	(SRP)	0.776	−0.276	0.014	0.144	0.005	0.699
Nitrite and nitrate	(NINT)	0.640	−0.325	−0.377	0.148	0.314	0.778
Ammonia	(NH)	0.838	−0.335	0.018	0.259	0.004	0.881
Reactive silica	(RSIO)	0.305	−0.008	0.769	0.212	0.157	0.754
Dissolved organic phosphorus	(DOP)	0.221	0.109	−0.030	0.955	−0.023	0.974
Total biomass	(TBIO)	0.231	0.038	−0.831	0.138	0.122	0.780
Cyanophyta	(CYAN)	−0.096	0.832	−0.093	0.021	−0.108	0.722
Chlorophyta	(CLOR)	−0.028	−0.064	−0.244	−0.054	0.915	0.904
Chrysomonadinae	(CHRY)	−0.129	0.779	0.182	−0.069	−0.180	0.693
Crypotomadinae	(CRYP)	0.069	0.733	0.001	0.115	0.110	0.567
Dinophycinae	(DINO)	−0.175	0.719	−0.210	−0.045	−0.525	0.869
Diatomeae	(DIAT)	0.315	−0.076	−0.699	0.187	0.337	0.742
Factor number		1.	2.	3.	4.	5.	
Factor Interpretation (This indicates which variables were correlated and constituted a factor)		+PRIP +PARP +TFP +SRP +NINT +NH	−CHLO +TEMP +CYAN +CHRY +CRYP +DINO	+TEMP +RSIO −TBIO −DIAT	+TFP +DOP	+CLOR −DINO	

nine seasonal correlations (spring, summer, and fall) between biomass and three of the soluble nutrients (soluble reactive silica, soluble reactive phosphorus, and soluble inorganic nitrogen), only two of the correlations were statistically significant. There are many explanations on hand that partially account for the lack of correlation of biomass with nutrients but none that are complete. These possible explanations are: a) high concentrations of soluble nutrients and biomass are found near river inputs which contrast with the low concentrations of nutrients found with biomass in the offshore environment; b) organic detritus is usually in much higher concentration in the water than live organic material (Leach 1975; Burns 1976) and detritus decomposition may control nutrient concentrations by regeneration to a greater extent than phytoplankton nutrient uptake; c) settling and turbulence processes can move the phytoplankton away from the water

TABLE 6. Factor analysis of 17 variables measured during the summer of 1970 in Lake Erie.

		Factor					
		1	2	3	4	5	
Sum of squares		4.977	2.447	2.207	2.068	1.982	
Variable		Factor Loadings					Communality
Chlorophyll	(CHLO)	0.941	0.124	−0.088	−0.048	−0.025	0.911
Primary production	(PRIP)	0.894	0.009	0.039	0.060	−0.119	0.818
Temperature	(TEMP)	0.210	0.129	0.019	0.580	−0.706	0.895
Particulate phosphorus	(PARP)	0.745	0.137	0.250	−0.178	−0.073	0.673
Total filtered phosphorus	(TFP)	0.185	0.881	0.352	−0.052	0.053	0.940
Soluble reactive phosphorus	(SRP)	0.477	0.196	0.665	−0.162	0.085	0.742
Nitrite and nitrate	(NINT)	0.180	0.561	0.603	−0.049	0.044	0.715
Ammonia	(NH)	−0.166	0.156	0.718	0.195	−0.230	0.659
Reactive silica	(RSIO)	−0.013	−0.142	0.492	0.697	−0.220	0.796
Dissolved organic phosphorus	(DOP)	0.043	0.982	0.001	0.012	0.038	0.968
Total biomass	(TBIO)	0.804	−0.076	0.058	0.297	0.414	0.915
Cyanophyta	(CYAN)	0.522	−0.052	−0.239	0.394	−0.542	0.782
Chlorophyta	(CLOR)	0.804	−0.140	−0.231	0.265	−0.035	0.791
Chrysomonadinae	(CHRY)	0.474	−0.453	−0.483	0.069	−0.063	0.672
Cryptomonadinae	(CRYP)	0.166	0.022	−0.235	0.021	0.823	0.762
Dinophycinae	(DINO)	0.099	0.057	−0.080	0.895	0.018	0.821
Diatomeae	(DIAT)	0.738	0.146	0.193	0.135	0.448	0.821
Factor number		1.	2.	3.	4.	5.	
Factor Interpretation (This indicates which variables were correlated and constituted a factor)		+CHLO +PRIP +PARP +SRP +TBIO +CYAN +CLOR +CHRY +DIAT	+TFP +NINT +DOP −CHRY	+SRP +NINT +NH +RSIO −CHRY	+TEMP +RSIO +DINO	−TEMP −CYAN +CRYP +DIAT +TBIO	

mass in which they actually grew. The lack of correlation between soluble nutrients and biomass is probably the result of a complex combination of these three phenomena. A study of the relationships between phytoplankton and soluble nutrients in the waters of Lake Erie requires further research by following the kinetics of growth, nutrient uptake and release, in a specific body of water.

Although primary production was seen to be correlated with temperature in the summer (Table 1), no significant correlation between these two variables was observed in the spring and fall due to the ability of certain diatom species to be productive at lower temperatures. Factor analysis revealed that temperature was never significantly correlated with primary production, suggesting that temperature is not an important controlling factor for primary production in Lake Erie although the importance of temperature in causing changes in species composition is well known.

The six variables discussed in this study have been drawn together in a recent phytoplankton model of western Lake Erie (DiToro et al. 1973) in which chlorophyll *a* was taken as being directly proportional to phytoplankton biomass. The expression used for estimating phytoplankton growth is typical and was formulated as follows:

$$G_p = K \cdot P \cdot r \cdot K_1(T) \cdot \frac{C_n}{K_n + C_n} \cdot \frac{C_p}{K_p + C_p} \quad (1)$$

where G_p = phytoplankton biomass growth
K = constant
P = phytoplankton biomass concentration
$= K_c C_a$

TABLE 7. Factor analysis of 17 variables measured during the fall of 1970 in Lake Erie.

		Factor					
		1	2	3	4	5	
Sum of squares		3.801	2.600	2.501	2.480	1.602	
Variable		Factor Loadings					Communality
Chlorophyll	(CHLO)	0.075	−0.169	−0.861	0.049	−0.196	0.817
Primary production	(PRIP)	0.153	−0.840	−0.337	0.074	−0.112	0.861
Temperature	(TEMP)	0.664	−0.294	0.082	−0.551	0.149	0.860
Particulate phosphorus	(PARP)	−0.636	−0.381	−0.246	0.306	−0.133	0.722
Total filtered phosphorus	(TFP)	−0.891	0.089	0.033	0.109	0.109	0.826
Soluble reactive phosphorus	(SRP)	−0.748	0.164	0.075	0.087	0.428	0.783
Nitrite and nitrate	(NINT)	−0.828	−0.031	−0.004	0.294	0.043	0.775
Ammonia	(NH)	0.085	−0.004	0.064	−0.211	0.814	0.719
Reactive silica	(RSIO)	−0.349	0.780	−0.092	0.221	−0.005	0.788
Dissolved organic phosphorus	(DOP)	−0.707	0.129	−0.034	0.059	−0.090	0.529
Total biomass	(TBIO)	−0.127	−0.412	−0.779	−0.179	−0.023	0.825
Cyanophyta	(CYAN)	0.330	−0.323	0.158	−0.652	0.174	0.694
Chlorophyta	(CLOR)	−0.094	−0.740	−0.067	−0.150	−0.078	0.590
Chrysomonadinae	(CHRY)	0.115	0.141	−0.063	−0.837	−0.237	0.794
Cryptomonadinae	(CRYP)	0.286	−0.289	−0.029	−0.799	0.121	0.819
Dinophycinae	(DINO)	0.163	−0.148	−0.098	−0.211	−0.731	0.637
Diatomeae	(DIAT)	−0.060	0.157	−0.953	0.096	0.007	0.945
Factor number		1.	2.	3.	4.	5.	
Factor Interpretation (This indicates which variables were correlated and constituted a factor)		+TEMP −PARP −TFP −SRP −NINT −DOP	−PRIP +RSIO −TBIO −CLOR	−CHLO −TBIO −DIAT	−TEMP −CYAN −CHRY −CRYP	+SRP +NH −DINO	

K_c = constant; C_a = chlorophyll *a* concentration
r = light limiting growth factor
$K_1(T)$ = temperature-dependent saturated growth rate
C_n, C_p = concentrations of soluble inorganic nitrogen and soluble reactive phosphorus, respectively
K_n, K_p = empirically determined constants

It is not possible to verify whether equation (1) is a correct formulation of the factors affecting phytoplankton growth in Lake Erie because every natural sample contains many different species of phytoplankton and it is probable that the constants in equation (1) are different for the different phytoplankton species and are also dependent on their size and physiological condition.

In light of the relationships investigated it is possible to comment on the variables included in equation (1) and on the limits of applicability of the equation.

The use of chlorophyll *a* as being representative of phytoplankton biomas must be undertaken with caution because it has been shown above that there is no relationship between these two variables in the spring. No comment can be made on r, the light limiting growth factor because this variable was constant during this investigation. However, the factor analysis showed that phytoplankton biomass, the soluble nitrogen nutrients and soluble reactive phosphorus were all positively correlated with primary production during one season or another. This indicates that these variables are probably growth-controlling factors at various times, confirming that these terms should be included in equation (1).

The temperature term, K_1 (T) has been related to the specific activity (doublings per day), u, by the expression, $u = 0.851\ (1.066)^T$ (Eppely 1972). In this expression, T is in the form of an exponent and thus changes in the value of T can cause large changes in the value of u. The activity coefficient (mg $C_{assimilated}$ per milligram biomass per day) is similar in nature to specific activity

and is often used in phytoplankton models. However, in this study, high values of the activity coefficient were observed at both high and low temperatures. This is in contradiction to the pattern predicted by Eppley's equation and it appears that the inclusion of the temperature term in equation (1) is questionable. It seems that the temperature term gives the upper limit of growth rate but that this limit is seldom achieved (Eppley 1972).

Summary and Conclusions

Samples were taken from the surface waters of Lake Erie during 10 surveys from April to December and analyzed for concentrations of phytoplankton biomass by species, soluble nutrients, and chlorophyll *a*. The primary production and temperature of the samples were also measured. The surveys were listed either as spring (April–June), summer (July–September), or fall (October–December) and the data grouped accordingly. The three resultant data sets were then subjected to factor analysis. The activity coefficients for the various primary production experiments were also calculated.

When the results obtained for each value at the different sampling stations were averaged on an annual basis and distribution maps were prepared, it was found that the distribution patterns were very similar, suggesting a positive correlation between the variables. However, detailed statistical analysis showed that this was not necessarily true. It was seen that chlorophyll *a* was not related to phytoplankton biomass in the spring but was related during the summer and fall. Primary production was similarly unrelated to biomass in the spring but was positively correlated during the summer and fall. However, no significant relationship on an annual basis, either direct or inverse, was found to exist between phytoplankton biomass and soluble nutrients. Primary production was found to be directly correlated with temperature in the summer only. The activity coefficient showed no relationship with temperature, with both high and low values for the coefficient being obtained under each of the different temperature conditions prevailing in the early spring, summer, and fall.

Factor analysis showed that phytoplankton growth was directly correlated only with nutrients in the spring but was covariant with nutrients, biomass, and chlorophyll *a* in the summer. In the fall, primary production was correlated with biomass and Chlorophyta but inversely correlated to reactive silica. This analysis also revealed that temperature was never covariant with primary production. In light of this finding and the observed independence of the activity coefficient from temperature, it would seem that the use of the temperature term in an expression to predict growth rates in Lake Erie would cause erroneous results.

Acknowledgments

We thank K. J.-M. Jaquet and C. K. Minns for their assistance in the statistical analysis of the data. Thanks are also due to J. E. Moore, I. F. Munawar, G. Dupuis, F. Rosa, and L. R. Culp for their assistance.

BURNS, N. M. 1976. Temperature, oxygen, and nutrient distribution patterns in Lake Erie, 1970. J. Fish. Res. Board Can. 33: 485–511.

CAMERON, E. M. 1967. A computer program for factor analysis of geochemical and other data. Geol. Surv. Can., Pap. 67-34: 42 p.

DITORO, D. M., D. J. O'CONNOR, J. L. MANCINI, AND R. V. THOMANN. 1973. A preliminary phytoplankton–zooplankton-nutrient model of western Lake Erie. *In* Systems analysis and simulation in ecology. Vol. 3. 592 p.

EPPELEY, R. W. 1972. Temperature and phytoplankton growth in the sea. Fish. Bull. 70: 1063–1085.

GÄCHTER, R., R. A. VOLLENWEIDER, AND W. A. GLOOSCHENKO. 1974. Seasonal variations of temperature and nutrients in surface waters of Lakes Ontario and Erie. J. Fish. Res. Board Can. 31: 275–290.

GLOOSCHENKO, W. A. 1975. Nutrient–primary production relationships in Central Lake Erie: a simple correlation approach. Ohio J. Sci. 75. (in press)

GLOOSCHENKO, W. A., J. E. MOORE, M. MUNAWAR, AND R. A. VOLLENWEIDER. 1974a. Primary production in Lakes Ontario and Erie: a comparative study. J. Fish. Res. Board Can. 31: 253–263.

GLOOSCHENKO, W. A., J. E. MOORE, AND R. A. VOLLENWEIDER. 1974b. Spatial and temperal distribution of chlorophyll *a* and pheopigments in the surface waters of Lake Erie. J. Fish. Res. Board Can. 31: 265–274.

JAQUET, J.-M., R. FROIDEAVAUS, AND J. P. VERNET. 1975. Comparison of automatic classification methods applied to lake geochemical samples. Math. Geol. 7: 237–266.

LEACH, J. H. 1975. Seston composition in the Point Pelee area of Lake Erie. Int. Revue Gesamten Hydrobiol. 60: 347–357.

LIND, D. T., AND R. S. CAMPBELL. 1969. Comments on the use of liquid scintillation for routine determination of C^{14} activity in production studies. Limnol. Oceanogr. 14: 878–889.

MUNAWAR, M., AND I. F. MUNAWAR. 1976. Lakewide study of phytoplankton biomass and its species composition in Lake Erie, April–December 1970. J. Fish. Res. Board Can. 33: 581–600.

STADELMANN, P., AND J. E. MOORE. 1974. Measurement and prediction of primary production at an offshore station in Lake Ontario. Fish. Res. Board Tech. Rep. 445: 12 p.

Seasonal Distribution and Abundance of Crustacean Zooplankton in Lake Erie, 1970

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WATSON, N. H. F. 1976. Seasonal distribution and abundance of crustacean zooplankton in Lake Erie, 1970. J. Fish. Res. Board Can. 33: 612–621.

The lakewide horizontal distribution and seasonal abundance of the cyclopoids, cladocerans, and calanoids in Lake Erie from April to December 1970, are presented.

Over much of the year cyclopoids dominated by *Diacyclops bicuspidatus thomasi* were most numerous with offshore centers of abundance. Several species of cladocerans, often with short peaks of occurrence, were most abundant during summer in the Western basin and associated with south shore population centers in the Central basin. Calanoids were frequently much less abundant than either of the two groups but present in the same areas as the cladocerans.

Most species present were short-lived with a high reproductive potential. Environmental clues are postulated to control specific abundance making group numbers more constant. Several life-cycle patterns were evident including species with spring–fall maxima and those with a single peak.

Timing of peak abundance of certain species and groups varies from west to east in the same fashion as temperature differences giving rise to east–west gradients of abundance which change with season. Cladoceran and calanoid groups show similarities in distribution pattern with estimates of particulate organic material rather than with algal abundance resulting in north–south gradients of abundance.

WATSON, N. H. F. 1976. Seasonal distribution and abundance of crustacean zooplankton in Lake Erie, 1970. J. Fish. Res. Board Can. 33: 612–621.

Nous décrivons la distribution horizontale et l'abondance saisonnière des cyclopoïdes, cladocères et calanoïdes dans tout le lac Erié d'avril à décembre 1970.

Durant la majeure partie de l'année, les cyclopoïdes, surtout *Diacyclops bicuspidatus thomasi*, sont le groupe le plus abondant, avec centres d'abondance au large. Plusieurs espèces de cladocères, souvent avec de courts pics d'occurrence, ont un maximum d'abondance et été dans le bassin occidental et sont associées avec les centres de population de la rive sud dans le bassin central. Les calanoïdes sont souvent beaucoup moins abondants que l'un ou l'autre des deux groupes, mais se trouvent dans les mêmes régions que les cladocères.

La plupart des espèces présentes ont une courte durée de vie et un fort potentiel reproducteur. Nous examinons certains indices de l'environnement qui contrôleraient l'abondance des espèces et donneraient plus d'uniformité aux nombres d'individus formant un groupe. Plusieurs schémas de cycle biologique sont évidents, y compris, chez certaines espèces, des maximums de printemps et d'automne, et des pics d'abondance uniques.

Le temps de pics d'abondance de certains groupes et espèces varie d'est en ouest de la même façon que les différences de température, ce qui donne naissance à des gradients est-ouest d'abondance variant avec la saison. Les groupes de cladocères et de calanoïdes présentent certaines similarités de distribution avec celle d'estimés de la matière organique particulaire plutôt qu'avec l'abondance des algues, ce qui résulte en des gradients nord-sud d'abondance.

Accepted November 18, 1975 Accepté le 18 novembre 1975

SINCE 1967 the Great Lakes Biolimnology Laboratory at the Canada Centre for Inland Waters has been involved in studies of the distribution and abundance of zooplankton in the Laurentian Great Lakes (Patalas 1972; Watson 1974). This report describes the distribution and seasonal abundance of crustacean zooplankton in the three basins of Lake Erie in 1970.

Printed in Canada (J4071)
Imprimé au Canada (J4071)

Various components of the crustacean zooplankton of Lake Erie have been studied over the past 40 yr through the efforts of many workers including Chandler 1940; Andrews 1953; Hubschman 1960; Bradshaw 1964; Davis 1968, 1969. Few of these early reports discuss all taxonomic groups of the zooplankton on a lakewide basis except Davis (1968, 1969). However, his sampling periods were restricted to three, July and October 1967 and January 1968, along a transect

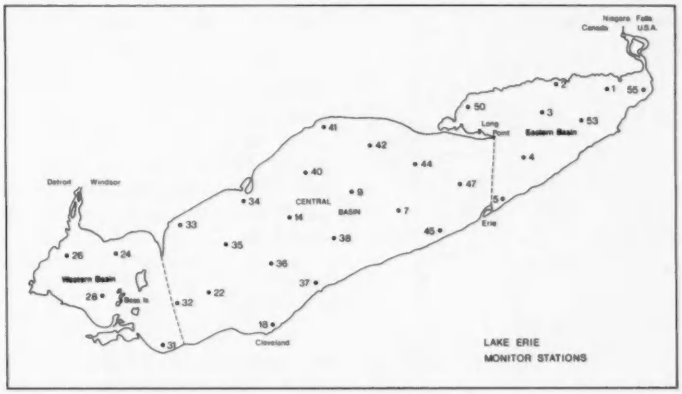

FIG. 1. Lake Erie showing stations sampled in 1970 for zooplankton species composition and abundance.

the length of the lake. Other reports are limited to the summer season or discuss abundance of limited groups in limited areas. This paper describes the seasonal abundance and horizontal distribution of the major crustacean zooplankton species and groups from April to December 1970 and relates seasonal patterns of abundance to water quality gradients.

Materials and Methods

Collections were taken from 30 stations distributed over Lake Erie (Fig. 1) on 10 synoptic cruises at time intervals of approximately 4 wk, from April to December 1970. Samples were taken from vertical net hauls of unmetered nets, with a mouth diameter of 40 cm and a mesh size of 64 μ. Net hauls were made according to the method of Patalas (1972) and Davis (1968, 1969) from 50 m to the surface or from 2 m above the bottom to the surface at stations with a depth of less than 50 m. The net hauls were concentrated on a 64-μ screen, washed into jars and preserved with 4% formalin.

In the laboratory several 1-ml subsamples were removed from each sample and counted with the aid of inverted microscopes until a total of at least 200 individuals had been enumerated. Adult calanoid and cyclopoid copepods were identified to sex and species. Cyclopoid copepodites were identified to species, juvenile calanoids were grouped to genus. Copepod nauplii and rotifers although numerous, were counted but not identified to species. Cladocerans were identified where possible to species and sex. Several samples from each cruise were subsequently reexamined to validate identifications of *Daphnia*. All sample results were converted to numbers per m^3 (1/liter = $1000/m^3$) assuming that the net strained all of a vertical column of water at the stations. Basin averages were calculated using the divisions shown in Fig. 1. Individual species data are on file by station and cruise.

Results

Species Composition

In general, the species of Crustacea present in the zooplankton samples are similar to those reported by previous workers (Davis 1968, 1969; Patalas 1972). Among the cladocerans, the daphnids were found to be highly variable in form. The forms recognized were *Daphnia retrocurva*, *D. longiremis*, and *D. galeata mendotae*. No forms of *D. ambigua* were recognized from the lake as mentioned by Patalas (1972), nor were any animals present in the 1970 samples which could be attributed to *D. pulex*.

Considerable variation was also noted in bosminids collected from Lake Erie. For example, the form attributable to the "cornuta" type of *Bosmina longirostris* was frequently collected in the samples. Similarly, a mucronate form with the rostral sensory bristle in a position similar to that in *Eubosmina coregoni* was found. Closer examination showed that this form has a postabdominal claw of the *B. longirostris* type, and these forms were designated *Bosmina* with mucro as no bosminids with an *Eubosmina* type claw (Deevey and Deevey 1971) and a true mucro were found in the samples.

Other species of cladocerans, *Chydorus sphaericus*, *Ceriodaphnia lacustris*, *Diaphanosoma leuchtenbergianum*, *Holopedium gibberum*, and *Leptodora kindtii* were seasonally important. On several occasions, species considered to be littoral in distribution, including *Alona affinis*, *Eurycerus lamellatus*, and *Latona setifer* were recorded at many stations.

Three suborders of copepods, *Calanoida*, *Cyclopoida*, and *Harpacticoida* were represented in the plankton. Calanoids present included *Limnoca-*

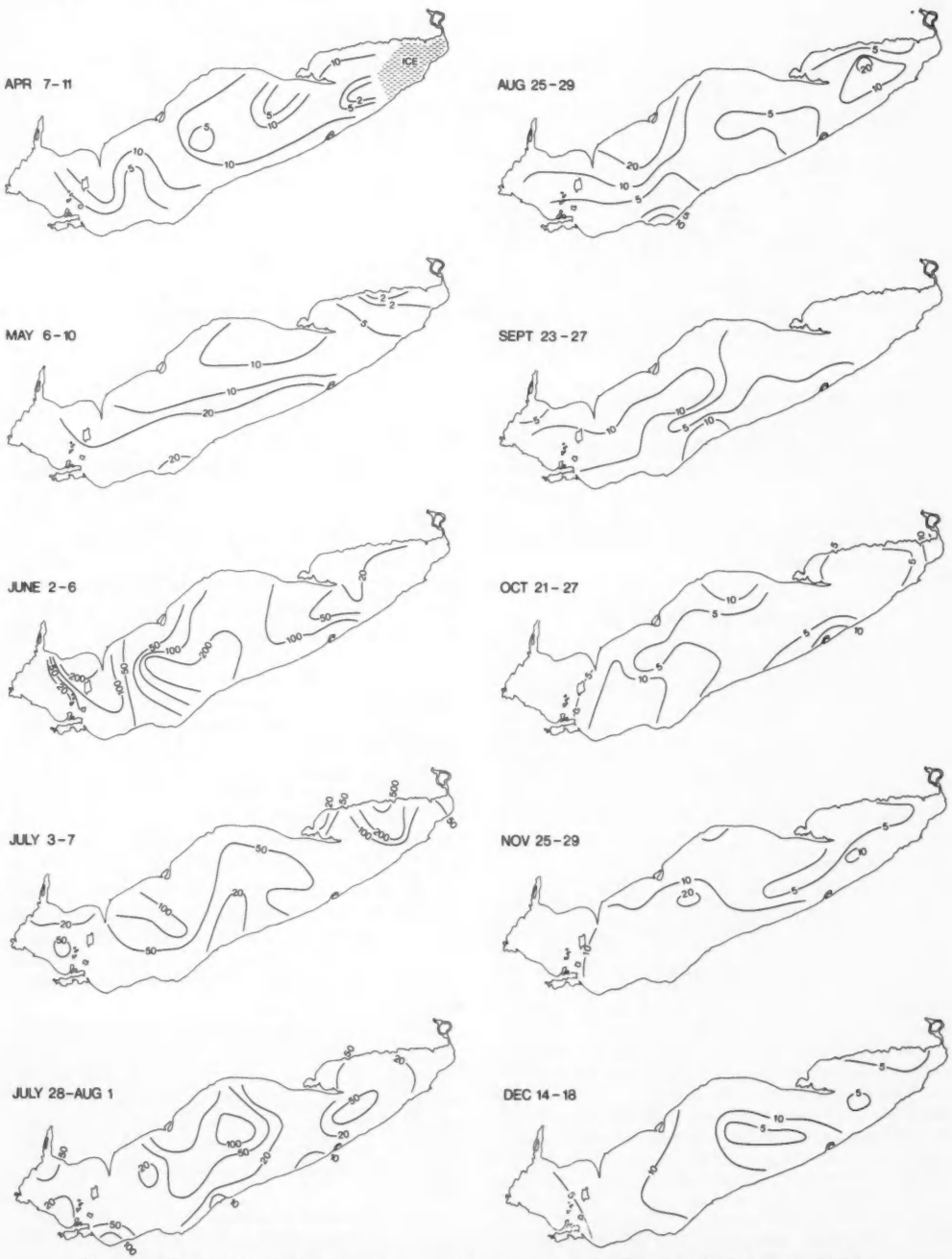

FIG. 2. Horizontal distribution of cyclopoid copepods, Lake Erie (April–December 1970).

lanus macrurus, *Eurytemora affinis*, *Epischura lacustris* and five species of *Diaptomus* — *D. oregonensis*, *D. sicilis*, *D. minutus*, *D. siciloides*, and *D. ashlandi*. Cyclopoid copepods included *Diacyclops bicuspidatus thomasi*, *Acanthocyclops vernalis*, *Mesocyclops edax*, *Tropocyclops prasinus mexicanus*, and a few isolated *Eucyclops speratus*. A few harpaticoids, assigned to the genus *Canthocamptus* were found at many of the inshore stations (less than 10 m) in all basins but were never numerically important.

Distribution and Abundance

Distributional maps (Fig. 2) show more uniform distributions of cyclopoids than other groups, with maximum values widely spread throughout all basins. There is only slight evidence of areas of local influence or distribution for this group. Centers of highest abundance often appeared to be in the open waters of the basins rather than inshore.

Cyclopoids (adult and copepodites) were often the most abundant group of crustacean zooplankton at many stations over much of the sampling period. Maximum numbers and sustained levels were similar in all three basins although timing of maxima was somewhat different (early June, Western and Central basins; early July, Eastern basin) (Fig. 3A).

Numbers over most of the year were about 10/liter in all three basins except for the summer reproductive pulse when maxima of 200/liter occurred.

Four species of cyclopoids were present in appreciable numbers, *(Dia)cyclops bicuspidatus thomasi*, *(Acantho)cyclops vernalis*, and *Mesocyclops edax* were most abundant while *Tropocyclops prasinus mexicanus* was far less numerous.

(Dia)cyclops b. thomasi was frequently the most abundant adult cyclopoid in the lake, although in the Western basin its abundance was surpassed by *A. vernalis* and *M. edax* in June and July. The greatest abundance of *D. b. thomasi* adults and copepodites occurred by late July. Both a spring and an early summer pulse of adults (Fig. 3B) and copepodites (Fig. 3C) is evident in the Western and Central basins with an apparent single pulse in the Eastern basin. During the spring and early summer period this species frequently made up between 60–80% of all individual at many stations in the Central and Eastern basins. Midsummer values of copepodites are lowest in all three basins with some indication of a late fall pulse developing in all basins. The midsummer decline in numbers of copepodites to 1/liter is interpreted as a loss of animals to the sediments where they may remain in arrested development.

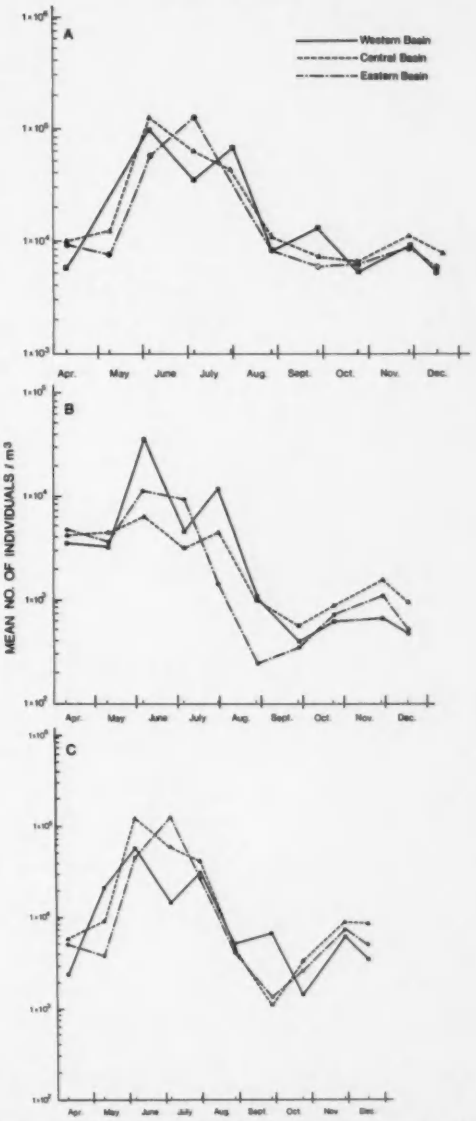

FIG. 3. Abundance of A) Cyclopoid copepods; B) Adult *(Dia)cyclops bicuspidatus thomasi*; and C) copepodite *D. b. thomasi*.

(Acantho)cyclops vernalis appears to have three main pulses of adults and copepodites in the Western basin where it was most abundant (maximum 20/liter); in June (following the maximum abundance of *D. b. thomasi*), in September, and in early winter. Numbers of adults and copepodites are low and fairly consistent

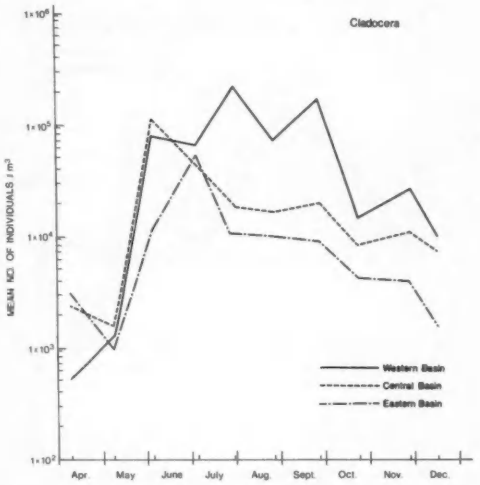

FIG. 4. Abundance of cladocerans by basin (April–December 1970).

(2/liter) from June to December in the Central basin and occurrences are somewhat sporadic in the Eastern basin.

Mesocyclops edax appears to have three pulses in the lake. The first peak of copepodites preceeded appearance of adults and the final copepodite peak has no corresponding adult peak in the late fall. It is, therefore, inferred that the species overwinters as copepodites. Declining numbers encountered in the fall and low numbers in the early spring indicate before the presence of adults that these animals may enter the sediments as well. Although widely distributed through the lake (at low numbers less than 1/liter), *T. p. mexicanus* was most abundant in the Western basin, especially during the summer and fall months.

Seasonal abundance of cladocerans, the second most abundant group of crustacean zooplankton, especially in the Western basin, is shown in Fig. 4. In April and May, numbers were low in all three basins and increased rapidly by June in all three basins, with warming temperatures. Maximum numbers and sustained abundance over the summer were highest in the Western basin and least in the Eastern basin. Several pulses of abundance were evident in the Western basin associated with a succession of dominant forms. Seasonal distributional maps (Fig. 5) demonstrate the frequently patchy distribution of the group. Often, high values were associated with south short centers of population. Parallels can best be drawn with the distribution of particulate phosphorus (Burns 1976).

Seasonal distributions for most species are highly variable. Examination of the distribution generally indicates a single summer pulse for each species in each basin. An exception to this occurs in the seasonal abundance of the two bosminids encountered in the lake. Combined abundance of the several forms of mucronate-shelled *Bosmina* referred to as *B. longirostris* showed a bimodal distribution with an early June and a late November peak in the Western and Central basins and a midsummer minimum. In the Eastern basin there was a single early July peak followed by a steep decline to low numbers for the remainder of the season. The seasonal abundance of *Eubosmina coregoni* consisted of a broad summer peak from June to September in the Western basin and a bimodal pattern in the Eastern and Central basins. Numbers of this species declined considerably from west to east in the lake on almost every cruise.

Although concentrations of the two species of bosminids frequently were high, they were at times surpassed by those of *Daphnia* spp., especially in the Central and Eastern basins. Numbers rose rapidly to a peak in July and August and declined thereafter. Western basin abundances frequently exceeded those in the Central and Eastern basins until after September when they dropped more rapidly than elsewhere.

Three species of Daphnia were recognized in our Lake Erie samples, *D. longiremis*, *D. retrocurva*, and *D. galeata mendotae*. As there was considerable difficulty in identifying the species of *Daphnia* during routine counts, subsamples of the daphnids of some samples were examined for specific identification. These results indicate *D. longiremis* was most abundant in the eastern end of the lake and earlier in the year; *D. retrocurva* was more abundant in the Western basin earlier on; and *D. g. mendotae* was most abundant lake-wide after August.

Among less frequently occurring cladocerans, *Chydorus sphaericus* was most numerous in the Western basin with a maximum abundance in August of all three basins. Both *Leptodora* and *Diaphanosoma* were most abundant in the Western basin while *Holopedium* was most abundant in the Eastern basin. *Ceridaphnia* was found sporadically, most frequently in the Eastern basin.

Calanoids were the least abundant group of crustacean zooplankton and showed the least fluctuation in numbers (Fig. 6). Numbers were practically identical at all times in the Central and Eastern basins and were considerably more numerous in the Western basin until late July. Distributions were frequently highest along the south shore and showed localized concentrations associated with population centers (Fig. 7).

Diaptomus minutus and *D. oregonensis* were the most abundant calanoids in the lake es-

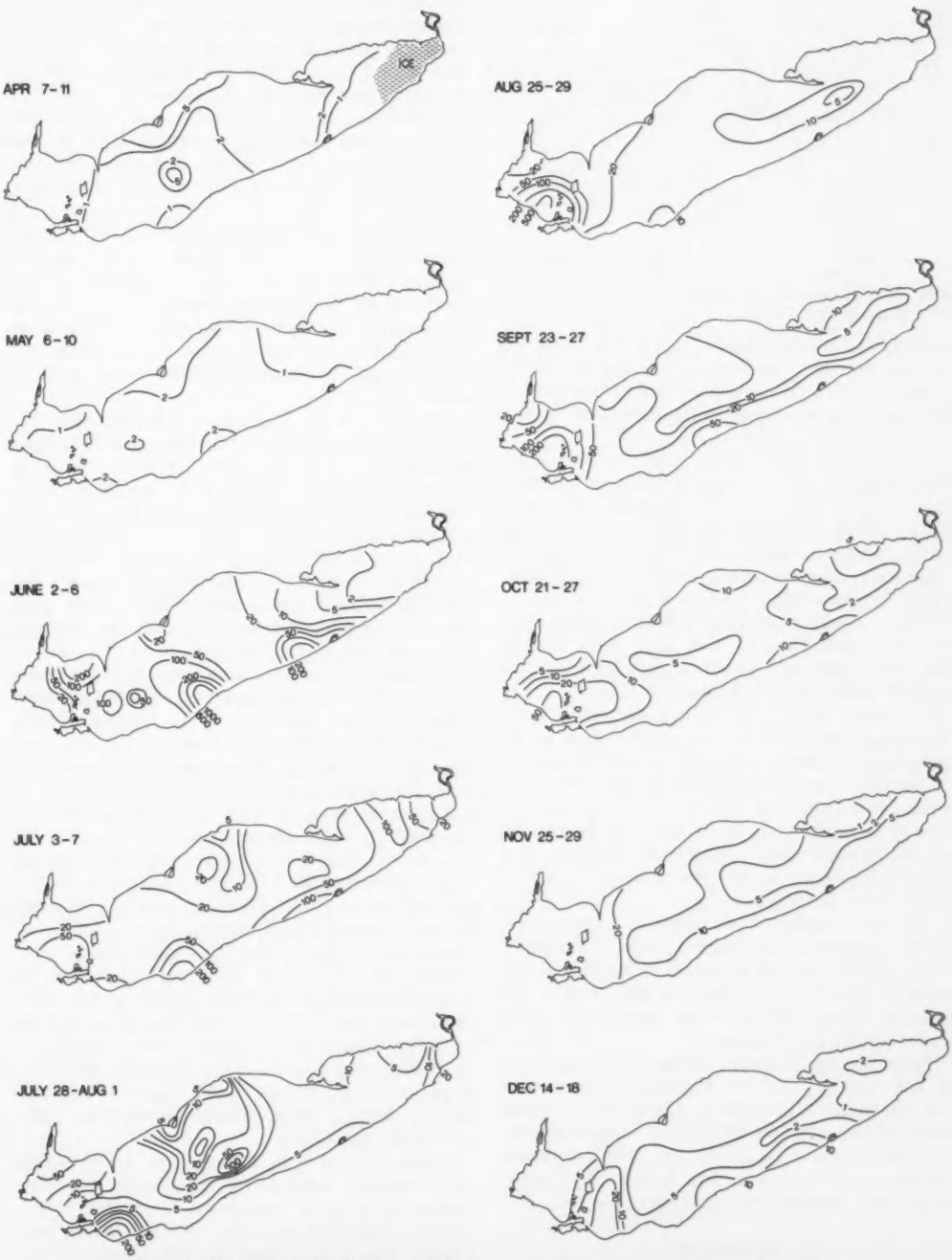

FIG. 5. Horizontal distribution of cladocerans, Lake Erie (April–December 1970).

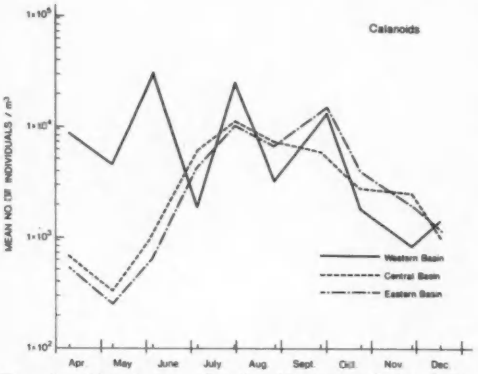

FIG. 6. Abundance of calanoid copepods by basin (April–December 1970).

pecially the Western basin. Numbers were highest in the Western basin for both species, April, August, and in the rest of the lake from June to October, perhaps related to the pattern of heating and cooling in the lake.

Diaptomus minutus adults were the most abundant calanoid in the Western basin of the lake until August. There is some evidence of several pulses of adults during the season. Among the other calanoids, *D. oregonensis* showed an inverse pattern of abundance to that of *D. minutus* in the Western basin compared with the rest of the lake. Adults were most abundant in the Central and Eastern basin (2.5.1/liter) from June to December, but most abundant (at about 1/liter) in the Western basin from April to June.

Diatomus sicilis and *D. siciloides* were found throughout the year in very low numbers in all three basins, showing a trend to decline through the season. *Diatomus siciloides* occurred in somewhat fewer numbers than *D. sicilis*, most frequently in the Western basin and least frequently in the Eastern basin. *Diaptomus ashlandi* was found infrequently in all three basins, most abundantly in the Central basin.

Average numbers of calanoid copepodites, mostly *Diaptomus* ranged from 1.0 to 10.1/liter on most cruises in all three basins except for lower numbers from April to June in the Central and Eastern basins. No obvious associations with adult abundances is apparent.

Three species of larger calanoids were present in low numbers in the plankton. *Eurytemora* was the most numerous and was found most frequently in the Western basin. *Limnocalanus* was found most frequently in Western basin samples and *Epischura* was found only infrequently and restricted to the Western basin.

Discussion

The lakewide distribution and abundance of crustacean zooplankton described in this paper indicates considerable horizontal variation on any one cruise and marked seasonal variation throughout the year in the abundance of any particular species and for group totals. This is to be expected from the work of Hubschman (1960) who found considerable day-to-day variation in crustacean zooplankton abundance at a station in the Western basin. Similarly, our distribution maps indicate that basin means may give a misleading picture of lakewide conditions because distributions are often far from uniform across a basin.

In light of what must be rapidly changing temporal abundances at any location in the lake, it is very difficult to describe in anything but the most general way, relationships of the crustacean zooplankton to other environmental conditions at that locality. For this reason, major group responses will be examined rather than specific ones.

Patalas (1972) suggested from lakewide surveys that crustacean numbers peaked in mid-June and declined in mid-August as found in the present study (see also, Watson and Carpenter 1974). Patalas considered cruise mean estimates of zooplankton numbers in the Great Lakes, except Lake Erie, were positively related to chlorophyll *a* concentration and to heat content as well. He felt this relationship broke down in Lake Erie because temperatures were above the optimal range for zooplankton production or that factors other than food supply were limiting. This study was coupled with measurements of many more parameters and extended over a larger season than Patalas (1972). Thus it should provide confirmation of Patalas' hypothesis or relationships with other parameters.

Glooschenko et al. (1974) demonstrated that estimates of zooplankton abundance (from coulter counts) are not in phase with the seasonal cycle of chlorophyll *a* or phaeopigments. Distributional maps of zooplankton groups presented in this paper are either unrelated (cyclopoids) or frequently inversely related (cladocerans, calanoids) to chlorophyll abundance presented in Glooschenko et al. over the surface of the lake. However, examination of the distribution maps of other environmental factors from Gächter et al. (1974) and Burns (1976) and seasonal abundance of each of the three major crustacean groups indicate some general relationships with environmental factors.

There appears to be some general relationships with crustacean abundance and temperature. The timing of peaks of abundance of cyclopoid and calanoid copepods is earliest in the Western and Central basins and later in the Eastern basin related to rates of warming and cooling. The

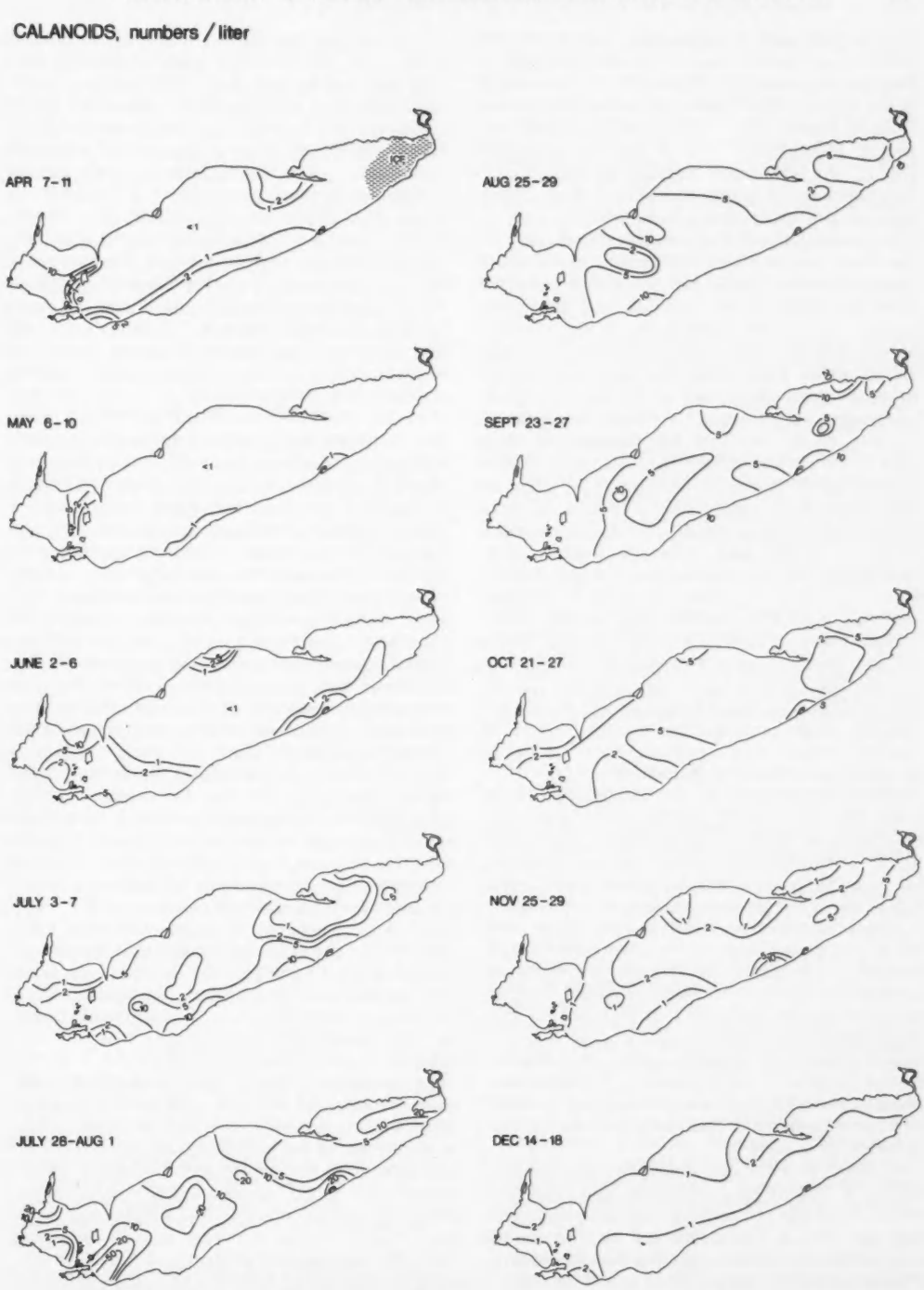

FIG. 7. Horizontal distribution of calanoid copepods, Lake Erie (April–December 1970).

single major peak in cladoceran numbers in the Central and Eastern basins (made up largely of *Bosmina longirostris*) (Figure 4) occurs earlier in the Central than Eastern basins as well. In the Western basin, where cladoceran abundance has several pulses each with a different dominant species, the abundance through the season follows the seasonal temperature curve more closely than in the other two basins. Patalas' (1972) observations that total crustacean abundance (for his study, primarily in the Central and Eastern basins) does not follow heat content is explained from this study by the fact that total abundance in these two basins is dominated by the numbers of two species, *Diacyclops bicuspidatus thomasi* and *Bosmina longiristris*. The major population peaks of these species are in the early summer, and resting stages occur later when temperatures reach a maximum. That lake temperature does play a role in the timing of these peaks, either directly on development, or through an effect on food availability, can be shown because the peak abundance for each species in the Eastern basin comes a month later than the Central basin maximum. Surface temperatures on the Eastern basin lag a month behind those in the Central basin as well. Perhaps the other species which replace them in the Central and Eastern basins do not have sufficient reproductive potential to sustain high population values, whereas in the Western basin pulses of cladocerans keep total numbers high until cooling of the waters in October brings concentrations down close to levels found in the other two basins.

The basic structure of the zooplankton in the Lake Erie community appears to be pulses of one, two or more generations of short-lived species with high reproductive potential building up to numbers governed by environmental factors. Their numbers then decreased as conditions trigger production of resting stages. Thus, one would not expect seasonal abundance of a single species to be directly related to environmental factors like food or temperature while that of the major groups might be. It is probably for this reason that abundances of cladocerans and calanoids appear to delineate zones of influence around population centers and to levels of particulate phosphorus (as a measure of detrital food) better than do total zooplankton or data or individual species.

It should be noted also that these opportunistic species of cladocerans, cyclopoids, and even calanoids, dominate the zooplankton community, all year, not just in the spring and summer as in Lake Ontario or in the summer in Lake Huron (Watson and Carpenter 1974).

Therefore, the seasonal abundance pattern becomes more similar to that of a shallow enriched pond than the relatively stable long-term communities of the upper lakes which have a dominant component of large-bodied, long-lived forms.

Factors which control production of and allow release of resting stages (like daylight) become important in determining species succession as factors controlling growth rate (like temperature) and reproductive potential (like food). Watson and Smallman (1971) determined that combinations of daylight and temperature initiated and terminated an arrest of development in the fourth copepodite instar in a pair of pond-dwelling cyclopoid copepods. From an analysis of populations in this study, the same mechanism is postulated to act as a control on reproductive activities and populations of *D. b. thomasi* and *M. edax* in Lake Erie. Stross and Hill (1968) have demonstrated similar temperature and photoperiodic control for the production and hatching of ehippia, the resting stage in cladocerans. It is, therefore, possible that closely related species with somewhat differing responses to environmental clues can share a particular resource temporarily without severely competing.

The similarities between cladoceran, and to a lesser extent, calanoid distributions to particulate phosphorus distribution rather than to chlorophyll *a* or phaeophytin distributions is noteworthy. It appears that the abundance of these forms is responding to available food sources in the form of organic detritus rather than to phytoplankton abundance. Gliwicz (1969) has suggested that in eutrophic waters the detritivore zooplankton food chain is favored over the herbivorous one of oligotrophic and mesotrophic waters. Particulate seston frequently exceeds algal biomass by considerable amounts in most lake studies, and concentrations of particulate phosphorus noted around population centres (Burns 1976) must indicate large sources of organic input to Lake Erie from domestic and industrial sewage as suggested by Leach (1975). This in turn must provide tremendous impetus for heterotrophic bacterial growth as well as a direct food source for the masses of microcrustacea that can develop at localized areas on the lake.

Diacyclops b. thomasi predominates the offshore plankton of the lake early in the year; its abundance indicates a tremendous food source at that time of the year, see McQueen (1969) for estimates of consumption. Highest abundances were found in areas of the Central basin overlying regions where surficial sediments have been anoxic during the past several summers. Thus the high organic content and biological activity of this anoxic zone of sediments appears to be producing conditions conducive to exploita-

tion by opportunistic species of relatively small size and short life cycle.

This analysis has not turned up the exclusion of any planktonic species other than *Limnocalanus* from large areas of the Central basin due to anoxic conditions in the restricted hypolimnion, although vertical net hauls are not the best method for obtaining such information. Gannon and Beeton (1971) tabulate data demonstrating a general decline of *Limnocalanus* in the lake as a whole, which they attribute both to eutrophication and fish predation. It seems unlikely that cold stenotherms like *Limnocalanus*, *Diaptomus ashlandi*, *D. sicilis*, and *Mysis relicta* could survive as resident populations in the Central basin at the present time.

Summary and Conclusions

Abundance of crustacean zooplankton during 1970 was marked by rapid changes in species number and composition at most of the stations in each basin. In general, maximum concentrations of individuals occurred in the shallow Western basin and minimum values occurred in the Eastern basin, however, on some cruises, minimum numbers were found in the Western basin. North–south gradients, (often associated with extremely high numbers of individuals, especially of cladocerans, around population centers on the south shore were as frequent as the east–west gradients described by other authors. These east–west gradients seemed to be associated with temporal lags in build-ups of species numbers related to basin differences in temperature.

The dynamics of the changing population numbers of a particular species are very difficult to assess because of the wide spacing of the samples in time and space.

For some species, especially among the cyclopoids, a seasonal pattern related to life-history events, can be discerned. Abundance of the cladocerans and warmwater calanoids with short generation time can be regarded as responses to environmental stimuli especially temperature and detrital food sources.

Acknowledgments

This study was initiated by Dr A. Nauwerck and Dr R. A. Vollenweider. Many people, especially Mr G. F. Carpenter and Mrs L. Devey, the technical operations crew of the M.V. *Martin Karlsen* contributed to the collection and analysis of the samples on which this study is based. The stimulus of Dr N. M. Burns and the remainder of the "Lake Erie group" has added materially in the preparation of this paper.

ANDREWS, T. F. 1953. Seasonal variations in relative abundance of *Cyclops vernalis* Fischer, *Cyclops bicuspidatus* Claus, and *Mesocyclops leuckarti* Claus in Western Lake Erie, July, 1946 to May, 1948. Ohio J. Sci. 53: 91–100.

BRADSHAW, A. S. 1964. The crustacean zooplankton picture — Lake Erie 1939–49–59: Cayuga Lake 1910–1951–1961. Verh. Int. Ver. Limnol. 15: 700–708.

BURNS, N. M. 1976. Temperature, oxygen, and nutrient distribution patterns in Lake Erie, 1970. J. Fish. Res. Board Can. 33: 485–511.

CHANDLER, D. C. 1940. Limnological studies of Western Lake Erie. I. Plankton and certain physico-chemical data of the Bass Islands region from September 1938 to November 1939. Ohio J. Sci. 40: 291–336.

DAVIS, C. C. 1968. The July 1967 Zooplankton of Lake Erie. Proc. Conf. Int. Assoc. Great Lakes Res. 11: 61–75.

1969. Seasonal distribution, constitution and abundance of zooplankton in Lake Erie. J. Fish. Res. Board Can. 26: 2459–2476.

DEEVEY, E. S., AND G. B. DEEVEY. 1971. The American species of *Eubosmina* (Crustacea, Cladocera). Limnol. Oceanogr. 16: 201–218.

GÄCHTER, R., R. A. VOLLENWEIDER, AND W. A. GLOOSCHENKO. 1974. Seasonal variations of temperature and nutrients in the surface waters of Lakes Ontario and Erie. J. Fish. Res. Board Can. 31: 275–290.

GANNON, J. E., AND A. M. BEETON. 1971. The decline of the large zooplankton *Limnocalanus macrurus* (Copep. Crust.) in Lake Erie. Proc. Conf. Great Lakes Res. 14: 27–38.

GLIWICZ, Z. M. 1969. Studies on the feeding of pelagic zooplankton in lakes of varying trophy. Ekol. Pol. Ser. A 17: 663–708.

GLOOSCHENKO, W. A., J. E. MOORE, AND R. A. VOLLENWEIDER. 1974. Spatial and temporal distribution of chlorophyll *a* and phaeopigments in surface waters of Lake Erie. J. Fish. Res. Board Can. 31: 265–274.

HUBSCHMAN, J. H. 1969. Relative daily abundance of planktonic crustacea in the island region of Western Lake Erie. Ohio J. Sci. 60: 335–340.

LEACH, J. H. 1975. Seston composition in the Point Pelee area of Lake Erie. Int. Rev. Gesamten Hydrobiol. 60: 347–356.

MCQUEEN, D. J. 1969. Reduction of zooplankton standing stocks by predaceous *Cyclops bicuspidatus thomasi* in Marion Lake, B.C. J. Fish. Res. Board Can. 26: 1605–1618.

PATALAS, K. 1972. Crustacean Plankton and eutrophication of the St. Lawrence Great Lakes. J. Fish. Res. Board Can. 29: 1451–1462.

STROSS, R. A., AND J. HILL. 1968. Photoperiod control of winter diapause in the freshwater crustacean, *Daphnia*. Biol. Bull. (Woods Hole, Mass.) 134: 171–198.

WATSON, N. H. F. 1974. Zooplankton of the St. Lawrence Great Lakes — Species composition, distribution, and abundance. J. Fish. Res. Board Can. 31: 783–794.

WATSON, N. H. F., AND G. F. CARPENTER. 1974. Seasonal distribution and abundance of crustacean zooplankton and net plankton biomass in Lakes Ontario, Erie, and Huron. J. Fish. Res. Board Can. 31: 309–317.

WATSON, N. H. F., AND B. N. SMALLMAN. 1971. The role of photoperiod and temperature in the induction and termination of an arrested development in two species of freshwater cyclopoid copepods. Can. J. Zool. 49: 855–862.

The Fish Community in Lake Erie[1]

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LEACH, J. H., AND S. J. NEPSZY. 1976. The fish community in Lake Erie. J. Fish. Res. Board Can. 33: 622–638.

Lake Erie, because of its shallowness and the fertility of its drainage basin, is the most productive of the Laurentian Great Lakes. The fish community is diverse (114 species representing 24 families) and is dominated by mesothermal and warmwater species. Although commercial fish production has been fairly consistent (average annual yield of 23,000 metric tons since 1915) there has been a dramatic change in species composition during the last 150 yr. Lake sturgeon, lake trout, lake herring, lake whitefish, sauger, and blue pike have virtually disappeared. Commercial production of medium-priced species such as yellow perch and white bass has increased and lower-valued species such as rainbow smelt, alewife, and freshwater drum have become more dominant in the fishery. The cultural stresses deemed largely responsible for these changes are, in approximate order of effect, an intensive commercial fishery, nutrient loading, the introduction of nonnative species, tributary and shoreline restructuring, erosion and siltation, and the introduction of toxic materials. The effects of these stresses on the environment and species composition are discussed.

The current status and future of the remaining commercial species are considered in the light of past and present effects of the cultural stresses. The joint efforts of the United States and Canada to improve water quality and to control and reduce nutrient loading should lead to an improvement in the environment in the near future. Lake Erie is well suited for the production of valuable percid species but unless a strong effort is made to regulate exploitation the future of the commercial fishery is precarious.

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A cause de son peu de profondeur et de la fertilité de son bassin hydrographique, le lac Erié est le plus productif des Grands lacs laurentiens. La communauté ichtyologique est diversifiée (114 espèces représentant 24 familles) et est dominée par des espèces mésothermiques et d'eau chaude. Bien que la production commerciale de poisson ait été assez uniforme (rendement annuel moyen de 23,000 tonnes métriques depuis 1915), il s'est produit un changement dramatique dans la composition en espèces au cours des 150 dernières années. L'esturgeon de lac, le touladi, le cisco de lac, le grand corégone, le doré jaune et le doré bleu ont pratiquement disparu. La production commerciale d'espèces de valeur moyenne telles que la perchaude et le bar blanc a augmenté. Les espèces de moindre valeur telles que l'éperlan arc-en-ciel, le gaspareau et le malachigan sont devenues plus dominantes dans la pêcherie. Les contraintes culturelles considérées comme largement responsables de ces changements sont, à peu près dans l'ordre de l'effet produit, pêche commerciale intensive, charge de substances nutritives, introduction d'espèces exotiques, restructuration des tributaires et de la ligne de rivage, érosion et envasement, et introduction de matières toxiques. Nous analysons les effets de ces contraintes sur l'environnement et la composition en espèces.

Le statut actuel et futur des espèces commerciales qui demeurent est considéré à la lumière des effets passés et présents des contraintes culturelles. Les efforts conjoints des Etats-Unis et du Canada en vue d'améliorer la qualité de l'eau et de contrôler et réduire la charge de substances nutritives devraient, dans un avenir prochain, améliorer l'environnement. Le lac Erié se prête bien à la production d'espèces de percidés de valeur mais, sans un effort concerté en vue d'en régler l'exploitation, l'avenir de la pêche commerciale est précaire.

Accepted November 18, 1975 Accepté le 18 novembre 1975

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[1]Contribution No. 75–12 of the Ontario Ministry of Natural Resources, Fish and Wildlife Research Branch, Maple, Ont.

Printed in Canada (J4086)
Imprimé au Canada (J4086)

LAKE Erie shares with Lake Michigan the distinction of having the largest number of fish species of the Laurentian Great Lakes. Van Meter and Trautman (1970) list 138 species from Lake Erie and its tributary waters exclusive of the Detroit River. Of these, 114 species are native to the lake (Christie 1974). The diversity is remarkable when related to the total number of freshwater species in Canada which is only 177 (Scott and Crossman 1973). Of the 24 families represented in Lake Erie, Cyprinidae, Percidae, Catostomidae, Centrarchidae, and Ictaluridae contribute 74% of the species (Table 1). The community is a mixture of coldwater, mesothermal, and warmwater species with a predominance of warmwater and mesothermal types. Coldwater species are now few and are largely restricted to the deeper, colder areas of the lake.

Lake Erie is the most productive of the Great Lakes in terms of fish harvest. In all but 4 yr since 1914 Lake Erie has led the other Great Lakes in commercial landings. Lake Michigan's production has exceeded that of Lake Erie recently because of the increased alewife catch but Lake Erie remains foremost in value of harvest. Lake Erie had produced more than one half of the Great Lakes total in 12 yr since 1914 and more than one third in all but 3 yr. In addition, the lake has become increasingly important for its sport fishery.

Because of the importance of the fish community to the lake ecosystem and as a food and recreational resource, it is appropriate to include a brief account of fish and fishing in Lake Erie. We shall attempt to describe the community in relation to man's influence, past and present. Unfortunately our information is biased towards those species that have been commercially significant; however, without the fishery records there would be little data available. Sporadic production data for some species date back to 1867 but complete statistics are available only from 1914.

TABLE 1. Families of fishes native to Lake Erie (from Christie 1974).

	No. of genera	No. of species
Petromyzontidae (lampreys)	2	3
Acipenseridae (sturgeons)	1	1
Lepisosteidae (gars)	1	2
Amiidae (bowfins)	1	1
Clupeidae (herrings)	1	1
Salmonidae (trouts, whitefishes)	2	4
Hiodontidae (mooneyes)	1	1
Umbridae (mudminnows)	1	1
Esocidae (pikes)	1	3
Cyprinidae (minnows)	12	33
Catostomidae (suckers)	8	14
Ictaluridae (catfishes)	3	8
Anguillidae (eels)	1	1
Poeciliidae (livebearers)	1	2
Gadidae (cods)	1	1
Gasterosteidae (sticklebacks)	1	1
Percopsidae (trout-perches)	1	1
Aphredoderidae (pirate perches)	1	1
Percichthyidae (temperate basses)	1	1
Centrarchidae (sunfishes)	5	11
Percidae (perches)	5	17
Sciaenidae (drums)	1	1
Cottidae (sculpins)	2	4
Atherinidae (silversides)	1	1
Total	55	114

Historical Review of Man's Influence on the Ecosystem

If we consider that ecosystems are products of their history (Watt 1973), then a broad understanding of the Lake Erie ecosystem is dependent upon an examination of the past record of fishing and other practices that affected it so greatly. The fish resources of Lake Erie have changed dramatically during the past 150 yr. These changes, which have been reviewed recently by Regier et al. (1969), Hartman (1972, 1973) and Regier and Hartman (1973), are considered to have resulted from cultural stresses. The major stresses acting on the fish community have been identified by Regier and Hartman (1973) as: the commercial fishery, cultural eutrophication, the introduction of new species, tributary and shoreline restructuring, shoreline erosion and consequent siltation, and the introduction of toxic materials from industrial sources. It is the purpose of this section to provide a brief summary of the major stresses and to record the responses of the community to these perturbations.

ENVIRONMENTAL CHANGES

Man's first serious impact on the Lake Erie drainage basin was the clearing of forests and savannas in the 18th and 19th centuries. The gradual conversion of woodlands and prairies to farmland, which was almost complete by 1870, led to increased erosion of soil (Trautman 1957). Turbidity from runoff materials increased in streams, rivers, and inshore areas of the lake. Important spawning areas for several valued species were degraded and inshore stands of aquatic plants in shallow bays of the lake were altered or eliminated. Stream-spawners were cut off from traditional spawning grounds by the construction

of dams on tributaries. The draining and filling of marshland bordering the lake further reduced the spawning areas of shallowwater species such as sturgeon (*Acipenser fulvescens*) and northern pike (*Esox lucius*).

As human population in the drainage basin grew, effluents from industries and towns flowed into tributaries or directly into the lake in ever increasing amounts. Population increased from about 0.5 million in 1840 to 3 million in 1900 (Beeton 1969). By 1970 the basin had a population of 11.1 million of which approximately 10% lived in Canada (Sly 1976). The Canadian part of the basin still remains primarily agricultural.

The overall effect of man's activities in the watershed has been an acceleration of the ageing process. Lake Erie, by virtue of its shallow depth and fertile drainage basin, was the most productive of the Great Lakes long before European man inhabited its shores. The increased flow of nutrients into the lake through man's efforts has led to qualitative changes in phytoplankton composition and production of organic matter far in excess of that which can be efficiently utilized by the ecosystem. Much of this surplus production, together with allochthonous organic matter, settles to the bottom where its decomposition leads to oxygen depletion. This process releases nutrients back to the water, thereby compounding the enrichment problem. Very simply put, the system virtually chokes on its own prosperity. Turbidity increases due to the suspension of phytoplankton and detritus. The principal physiological effect of this eutrophication on fish is stress caused by reduction of dissolved oxygen in the water (Fry 1969). Species of fish that are more tolerant of low oxygen conditions in some or all life-history stages or that can survive pelagically in the epilimnion are favored by eutrophication. Hence, the process can bring about a gradual shift in species composition, usually from higher to lower valued types.

The rate of ageing of a lake may be measured by the rate of change in such criteria as total productivity, diversity of types of organisms, amount of dissolved solids, water transparency, sedimentation, and the relative quantity of high and low-value fish. The dramatic changes that occurred in the Lake Erie environment during the past half century are summarized briefly as (data from International Joint Commission 1969 unless indicated otherwise): a) an increase in total dissolved solids from 133 to 198 mg/liter; b) increases in concentrations of calcium, sodium plus potassium, sulphate, and chloride of 40, 83, 350, and 550% respectively; c) a fivefold increase in open lake ammonia-nitrogen and a threefold increase in total nitrogen between 1930 and 1958; in the Western basin, ammonia-nitrogen increased thirteenfold between 1930 and 1967; d) a fourfold increase in total phosphorus and a tenfold increase in soluble reactive phosphorus in the Western basin between 1942 and 1967; e) a threefold increase in sedimentation rate since 1850 (Kemp et al. 1974); f) mean loadings of organic carbon, nitrogen, and phosphorus to sediments have increased 5, 7, and 4 times, respectively, since 1850 (Kemp et al. 1974); g) an eightfold increase in mean loadings of mercury to sediments, primarily since 1950 (Kemp et al. 1974); h) a twentyfold increase in phytoplankton numbers and a shift in species dominance in the Western basin from oligotrophic to eutrophic types; i) an increase in zooplankton abundance and a change in species composition towards eutrophic types; and j) a decline in numbers and diversity of pollution-intolerant benthic organisms and an increase in abundance of pollution-tolerant species.

Associated with the overall increase in productivity of the lake there has been an increase in the oxygen demand of the sediments. This increase has led to serious environmental difficulties for benthic communities during periods of thermal stratification. Although rich in organic matter, the Western basin is too shallow to stratify except during prolonged periods of hot, calm weather. Severe oxygen depletion occurred in 1953 during a 28-day windless period which resulted in a 90% reduction in the nymph population of the dominant mayfly, *Hexagenia* (Britt 1955). Carr et al. (1965) found that in 1963 only 5 days of hot, calm weather were required to reduce dissolved oxygen below 3 mg/liter. *Hexagenia* has been virtually eliminated from the Western basin while numbers of other organisms have increased. Carr and Hiltunen (1965) observed a ninefold increase in Oligochaeta, a fourfold increase in Tendipidae and a sixfold increase in Gastropoda between 1930 and 1961.

Stratification in the Central basin is a regular summer occurrence. Low dissolved oxygen concentrations in the hypolimnion have been recorded back to the first survey of 1929 (Carr 1962). Dobson and Gilbertson (1972) calculated that the recent oxygen depletion rate (3.6 mg/liter per month) is more than double the rate estimated for 1929. They determined that a depletion rate of 3.0 mg/liter per month will produce zero oxygen in the bottom water of the Central basin (calculated on a stratification period of 110 days). This critical rate was exceeded in 1961 and zero oxygen levels have been observed frequently since that year.

Thermal stratification occurs regularly in the

deep Eastern basin but critically low concentrations of dissolved oxygen have not yet been reported. However, current loadings of organic carbon to the Eastern basin are estimated to be 5 times that of early colonial inputs (Kemp et al. 1974) and no doubt the oxygen depletion rate for that basin is increasing. Hypolimnetic oxygen concentrations of less than 30% saturation have been reported in recent summers (Burns 1976).

Qualitative changes in the benthos of the Central and Eastern basins have not been well documented. Benthic organisms in the Central basin are generally considered to be pollution tolerant, although some oligotrophic forms such as *Mysis relicta* and *Pontoporeia affinis* still exist in the eastern part of the Central basin and in the Eastern basin (Beeton 1969). Brinkhurst et al. (1968) found a gradation in species of oligochaetes from west to east which parallels trophic conditions. Pollution-tolerant species were found in the extreme western end of the lake and species associated with eutrophic conditions were present in the Western and Central basins. Oligotrophic forms, which are common in the upper Great Lakes, were found in the Eastern basin.

Changes in Fishing Methods

The following brief account of fishing methods is summarized from Regier et al. (1969) and Applegate and Van Meter (1970).

Commercial fishing in Lake Erie began in 1795 with a hook and line fishery at Presque Isle, Pennsylvania. Regular commercial fishing in western Lake Erie commenced around 1815 with seining for lake whitefish (*Coregonus clupeaformis*), sauger (*Stizostedion canadense*), walleye (*Stizostedion vitreum vitreum*), and smallmouth bass (*Micropterus dolomieui*) in Maumee Bay. By mid-19th century seining had peaked in number of units and was gradually replaced by fishing with pound and gillnets. Pound nets were first set in U.S. waters in 1850 and in Ontario waters in 1869. Gillnets were first set commercially in 1852 in eastern Lake Erie; their use spread westward gradually, reaching the western end of the lake by 1877. Ontario fishermen did not use gillnets regularly until after 1900. A trot-line fishery for bullheads and channel catfish (*Ictalurus spp.*) was commenced around 1850 in Ohio waters. Other gear used in the U.S. fishery included trapnets and fyke nets which were popular in the late 1800s. Trapnets and fyke nets were banned from Ontario waters until 1950.

Early gillnets were made with coarse cotton twine and were fished from rowboats or sailboats. By 1870 the heavy nets were replaced with finer and stronger cotton or linen mesh and by 1900 most gillnetters were using steam-powered vessels and gillnet lifters. Efficiency in taking lake herring (*Coregonus artedii*) was increased by setting (canning-up) gillnets at various depths near the surface. Extra deep gillnets or bullnets were used at certain times of the year when lake herring ranged throughout a considerable part of the water column. These nets were introduced in 1905 and were banned in U.S. waters by 1934 and a few years later in Canadian waters because of their efficiency in taking young whitefish.

Gillnets made with multifilament nylon twine were introduced in 1949. Catch efficiency was increased further because the twine was almost invisible in the water. Interest in harvesting the abundant rainbow smelt (*Osmerus mordax*) led to the adoption of the otter trawl in the 1950s. With this gear and electronic fish-finding equipment, fishermen were able to take advantage of the schooling behavior of smelt. Many of the large diesel tugs in the Canadian fishing fleet are now equipped for both gillnetting and trawling. Canadian landings have exceeded those of the United States since 1954 due largely to the trawl fishery for smelt and an efficient gillnet fishery for yellow perch (*Perca flavescens*) (Fig. 1). The reduction in U.S. commercial production since 1956 has resulted from a decrease in fishing effort due primarily to declines in available labor and investment returns.

Changes in Native Fish Stocks

Lake Erie has led the Great Lakes fishery in variety of fish species and has often equalled or surpassed the combined production of the other Great Lakes. The commercial fishery has utilized more than 35 species since it began and about 19 species have been significant in the landings at one time or another (Hartman 1973). Commercial production has averaged about 23,000 metric tons/yr since 1915 (Fig. 1). Annual production has ranged from highs of about 34,000 metric tons in 1915 and 1956 to lows of approximately 14,000 metric tons in 1929 and 1941. Although the lake has yielded a fairly consistent harvest of fish flesh, there has been a dramatic change in species composition. The highly valued lake trout (*Salvelinus namaycush*), lake herring, lake whitefish, sauger, and blue pike (*Stizostedion vitreum glaucum*) have virtually disappeared. There has been an increase in production of medium-priced species such as yellow perch and white bass (*Morone chrysops*) and lower-valued species such as rainbow smelt and freshwater drum (*Aplodinotus grunniens*) have become more dominant in the fishery.

A brief historical account of the more important species in the fishery follows.

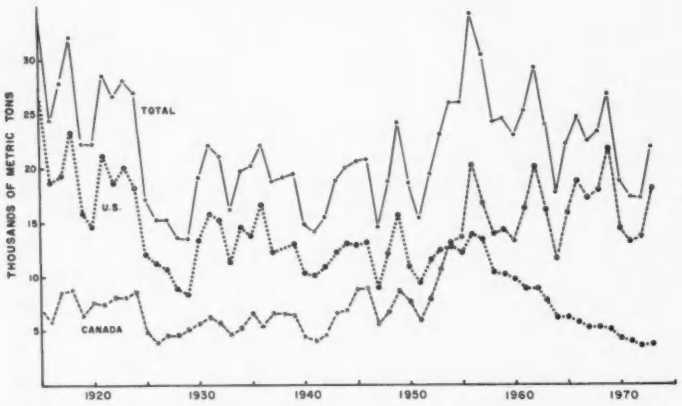

FIG. 1. U.S., Canadian, and total commercial fish production (all species) from Lake Erie, 1915–73 (from Baldwin and Saalfeld 1962 and Annual Summaries, Great Lake Fisheries, NOAA, National Marine Fisheries Service, Washington).

Lake sturgeon — This species, which was once abundant in the lake, is particularly vulnerable to exploitation due to its late maturity, slow growth, and ease of capture. In the early decades of commercial fishing, sturgeon were removed as a nuisance and used as fertilizer (Trautman 1957). By 1880 the sturgeon was recognized as an important commercial species and was exploited heavily. Catches declined after 1890 and by 1900 the species was scarce. The loss of river-spawning areas through damming hastened the decline of this species.

Lake trout — Distribution of lake trout, which is a slow growing, late maturing stenotherm, was restricted mainly to the Eastern Basin of Lake Erie. Annual catches exceeding 50 metric tons were taken in the late 1800s. After 1900 lake trout were of minor importance to the commercial fishery (Trautman 1957). Native lake trout are now extinct in Lake Erie. As Lake Erie is the southern extremity of lake trout distribution, this species preferred the deep water of the Eastern basin where it was particularly vulnerable to an intensive fishery. Moenig et al. (1974) considered that unrestricted exploitation was the single factor that eliminated this valuable resource from Lake Erie. Although changes in the lake basin were well under way by 1870, eutrophication did not have an appreciable impact on the environment in the Eastern basin until some time after the lake trout stock had been reduced significantly. Remnants of the population may have been reduced to extinction because of environmental degradation in spawning areas. It is not known whether the sea lamprey (*Petromyzon marinus*), which was first recorded in Lake Erie in 1921, hastened the extinction of this species.

Lake herring — The lake herring or cisco became a major commercial species in the 1880s. Commercial harvest fluctuated considerably, particularly between 1910 and 1925. Scott (1951) considered that periods of abundant commercial harvest resulted from dominant year-classes. The fishery collapsed in 1925 and production remained low until 1945 and 1946 when it recovered briefly and crashed again (Fig. 2). The species is now commercially extinct in Lake Erie.

Regier et al. (1969) considered that the 1925 decline of lake herring was mainly due to a too-intensive fishery. Canadian gillnetters were rapidly increasing their fishing effort at a time when stocks were under intensive pressure from the U.S. fishery. The destructive bullnets were fished extensively during this period. The fishery failed in the Western basin in 1920, in the Central Basin in 1925 and in the Eastern basin in 1929. Regier et al. (1969) suggested that the west-to-east progression in the collapse was evidence of excessive fishing pressure during the annual herring migrations. On the other hand, Beeton and Edmondson (1972) suggested that the west-to-east gradation was related to progressive eutrophication and degradation of habitat. We are inclined to agree with Regier et al. (1969), for if eutrophication was severe enough in the 1920s to detrimentally affect herring stocks, it is unlikely that the species would have recovered in the 1940s when environmental conditions had worsened.

The temporary recovery of the fishery in 1945 and 1946 was due mainly to the success of the 1944 year-class. Scott (1951) considered that the size of the spawning stock that produced this strong year-class must have been small since the commercial harvest in 1943 was insignificant and concluded that factors other than size of spawning stock were important in determining year-class strength. He was unable to positively relate other factors such as availability of food, predators, climatic conditions, disease, and siltation to year-class success. Regier et al. (1969) maintained that the ability of the remnant population to produce a strong year-class in 1944 did not seriously oppose their view that fishing was the principal factor in the collapse of lake herring in the 1920s.

Holling (1973) described the inability of the lake herring to rebound to previous levels of abundance even though fishing pressure was diverted to other species as a shift "from a domain with a high equilibrium to one with a lower one." He postulated that fishing pressure on lake herring and certain other Great Lakes stocks so reduced the resilience of the system that the introduction of an additional stress was sufficient to cause the population to crash. In other words, the populations were preconditioned for collapse by excessive harvesting.

Lake whitefish — This species is a cold stenotherm which requires silt-free spawning grounds for successful reproduction and cold, well-oxygenated, hypolimnetic waters for summer habitat. Lake Erie is the southern extremity of lake whitefish distribution.

By 1900 spawning runs into the Detroit River and Maumee Bay were curtailed by pollution and siltation (Trautman 1957) and the expanding fishery became dependent on lake-spawning stocks. Commercial harvests between 1913 and 1955 were characterized by periods of high and low production (Fig. 2). Lawler (1965) found that each peak in production was due to the presence of a particularly strong year-class which dominated the fishery for several years. He identified the 1926, 1936, and 1944 year-classes as major contributors to the fishery. The success of these year-classes could not be related to size of spawning populations or such factors as turbidity, wind, precipitation, and water level.

Lawler (1965) found that strong year-classes resulted only when water temperature was maintained close to the freezing point during development and hatching. In Lake Erie this meant that strong year-classes resulted only in those years when the spring rise in temperature was delayed 2–4 wk. There has been a general warming trend

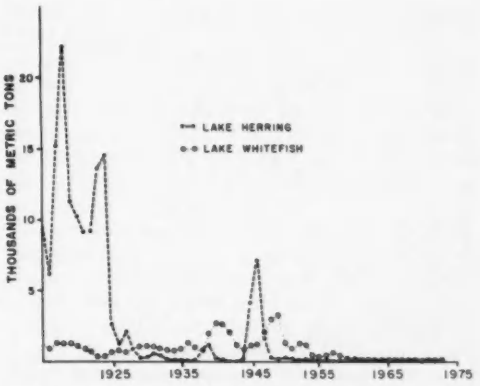

FIG. 2. Commercial production of lake herring and lake whitefish from Lake Erie, 1915–73.

in the climate of the Great Lakes basin which has resulted in an increase of 1.1 C in the mean annual temperature since 1918 (Beeton 1969). The Lake Erie environment is marginal for lake whitefish and Lawler (1965) concluded that if the lake continued to warm, the species would be increasingly confined to the deep waters of the Eastern basin and could become extinct.

Because of their high commercial value, whitefish have been fished intensively in Lake Erie and no doubt this has been partly responsible for the decline of the species. However, environmental degradation including siltation of spawning areas, declining oxygen levels in the Central basin hypolimnion, and increasing water temperature have all been effective in bringing about the decline of the species. Whitefish are occasionally taken in the Eastern and Western basins but could become commercially extinct.

Sauger — Commercial landings of sauger, one of three varieties of *Stizostedion* in Lake Erie, peaked in 1916 and decreased gradually to very low levels by 1956 (Fig. 3). Regier et al. (1969) considered that increased fertility and turbidity in the late 1800s benefited the light-sensitive sauger. They suggested that since the decline of the fishery after 1916 was fairly uniform, environmental conditions must have limited the success of the species. In particular, pollution, siltation, and damming may have degraded traditional spawning areas in inshore waters and tributaries of the south shore of the Western basin. But they also asserted that increasing fishing pressure must have been an important factor in preventing the population from maintaining abundance. The sauger inhabited primarily southshore waters of the Central and Western basins, perhaps due to preferred levels of

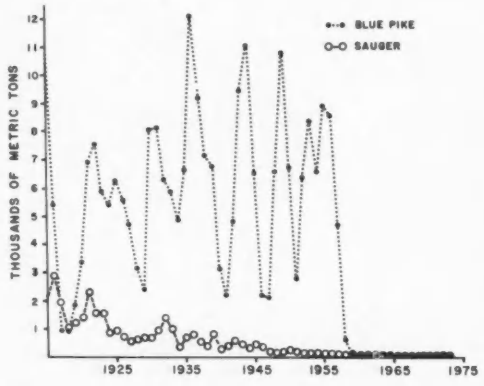

FIG. 3. Commercial production of blue pike and sauger from Lake Erie, 1915–73.

turbidity, and were taken mainly by Ohio fishermen. Native sauger is now commercially extinct in Lake Erie. Regier et al. (1969) speculated that introgressive hybridization with walleye may have been effective in eliminating the remnants of the sauger population.

Blue pike — Blue pike were abundant in the commercial fishery of the late 1800s but by 1915 landings began to fluctuate extensively. Production peaks in excess of 10,000 metric tons occurred in 1915, 1936, 1944, and 1949 and lows under 2,500 metric tons occurred in 1917–19, 1929, 1941, and 1946–47 before the fishery collapsed in 1958 (Fig. 3). The status of blue pike has been reported recently as follows: disappearance final (Regier et al. 1969); commercially extinct (Van Meter and Trautman 1970); rare, perhaps extinct (McAllister 1970); totally disappeared (Scott and Crossman 1973).

Regier et al. (1969) considered that an overintensive fishery, which disrupted self-stabilizing mechanisms within the population, led to the extreme fluctuations and ultimate crash of the fishery. Since young-of-the-year blue pike inhabited the same areas as older members of the population, they were vulnerable to cannibalism. Regier et al. (1969) postulated that overfishing for adults would cause unusual numbers of young-of-the-year to escape predation. This would lead to a short population explosion followed by several years of poor recruitment due to over-predation of abundant older fish on the young. An intensive fishery would cause increased amplitude in the fluctuations because the fish would be taken even when they were scarce. In addition, competition with and predation by the newly arrived rainbow smelt, which occupied the same habitat for part of the year, were likely detrimental to this species

The last successful year-class occurred in 1954 and there was virtually no recruitment to the fishery after that year (Parsons 1967). Production continued at high levels for another 3 yr and then collapsed. As growth rates in this period increased enormously, immature fish were readily exploited which further reduced spawning potential.

The reasons for the collapse of the fishery in 1958 have not been well defined. Summer oxygen deficiencies in the hypolimnion of the Central basin probably forced the blue pike into the deeper waters of the Eastern basin where they were more vulnerable to an intensive fishing effort. Regier et al. (1969) also suggested that introgressive hybridization with walleye may have been responsible for the final disappearance of the remnant stock.

Walleye — There are at least two discrete populations of walleye in Lake Erie, one largely confined to the Western basin and a smaller population resident in the Eastern basin. There is very little movement of adult walleyes between these two basins (Wolfert 1963) but Ferguson and Derksen (1971) observed a two-way movement of immature walleyes between the Western Basin and Lake St. Clair.

Before their usefulness was destroyed by pollution, siltation, and damming, many of the U.S. tributaries of the Western and Central basins were used by walleye for spawning (Langlois 1945). However, the environment remained suitable for lake spawners and the walleye has made an important contribution to the commercial fishery for many years. Annual harvests fluctuated narrowly around a mean of 825 metric tons between 1915 and 1936 and then started to increase steadily (Fig. 4). In the 1950s, landings climbed steeply to a peak of 7000 metric tons in 1962. Production has fluctuated at low levels since that date.

Walleye year-classes have become steadily more variable since the early 1950s (Hartman 1973) and Parsons (1970) has noted that they were generally strong in the period 1942–53 but substantially weaker in the subsequent interval 1954–61. As year-class strengths declined, growth rates increased, the average age of walleye taken by the fishery decreased, and more and more females were taken before they reached sexual maturity (Parsons 1970; Wolfert 1969). Most of the increase in growth rates occurred between the mid-1950s and the early 1960s. Present rates are considerably higher than those recorded during the 1940s (Wolfert 1972).

Regier et al. (1969) maintained that fishing intensity has been a major factor in the collapse of the walleye fishery. Ontario gillnetters greatly

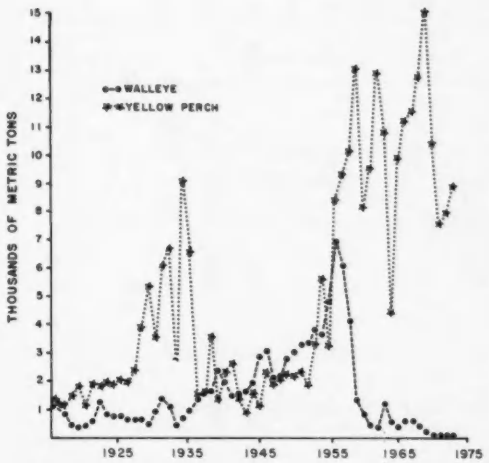

FIG. 4. Commercial production of walleye and yellow perch from Lake Erie, 1915–73.

increased the amount of gear starting in 1950 and their catches increased sevenfold by 1956. During this period fishing pressure by the Ohio fishery was also high.

Regier et al. (1969) considered that environmental degradation, which has led to alteration of bottom deposits and periodic regular oxygen depletion in the Western and Central basins, respectively, has reduced walleye habitat in summer. However, they do not consider that the demise of the mayfly population in the Western basin in 1953 had much effect on the walleye because it is a generalized predator. On the other hand, Beeton (1966) suggested that major changes in the environment and benthos in the mid-1950s may have had a major effect on the walleye population. Parsons (1970) also considers that the combination of environmental factors which resulted in oxygen depletion and the decline of the mayfly also led to the decline of the walleye.

Regier et al. (1969) have also postulated that pressure from the rapidly growing smelt population through competition and predation has had an additional deleterious effect on the walleye, particularly on fry carried by currents into the Central basin. However, smelt abundance is presently equal to if not greater than that of the mid-1950s, but does not appear to be affecting the success of recent year-classes of walleye.

Yellow perch — Yellow perch has been an important commercial species for many years with yearly production averaging about 2,600 metric tons between 1915 and 1950 (Fig. 4). During the 1930–36 period landings averaged 5,700 metric tons as fishermen diverted effort to the capture of perch after the decline of lake herring. As the harvest of the high-valued whitefish and blue pike declined in the 1950s yellow perch became even more important to the fishery. Production climbed to a peak of 15,286 metric tons in 1969 but has declined in recent years to a level of about 9,000 metric tons.

Whereas in the period from 1920 to 1950 U.S. fishermen had landed the greatest proportion of perch, Canadian fishermen increased production in the 1950s and have continued to be prime producers of yellow perch. Mean annual production from Canadian waters from 1948 to 1959 was 3,300 metric tons; between 1960 and 1973 production had increased to 8,500 metric tons. The effect of this increase in exploitation on the current status of the yellow perch resource will be discussed in the next section.

Current Status of Major Species

YELLOW PERCH

Current instability in the populations of yellow perch can be attributed to a variety of factors. As noted in the previous section, exploitation of yellow perch stocks has intensified since the decline of blue pike and walleye in the 1950s. Recently, partial and complete fishing bans on white bass and walleye because of high levels of mercury contamination have also tended to increase exploitative effort on this percid.

Greatest production during 1960–73 has been from Canadian waters (Fig. 5). Relative average contributions from the various districts to the entire Canadian catch have been: Western basin (District OE-1), 25%; west-Central basin (District OE-2), 49%; Central basin (District OE-3), 17%; east-Central basin (District OE-4), 3% and the Eastern basin (District OE-5), 5%. Production from these districts has fluctuated in the past with a gradual increase evident in the early 1960s, reaching the highest production ever in 1969. Although landings have shown a slight increase each year from 1971 to 1973, the figures are still well below the 1965–70 average of 10,300 metric tons. During the late 1950s and 1960s catch per unit effort (CUE) in the Western basin was high when effective effort.[2] was low. More recently, however, effort has increased in both the Western and Central basins (Great Lakes Fshery Commission

[2]Effective effort refers to the total gillnet effort expended which includes large and small gillnet mesh. All the gear lifted by a fisherman on a particular day is charged to a species taken, even if the amount is as little as a kilogram.

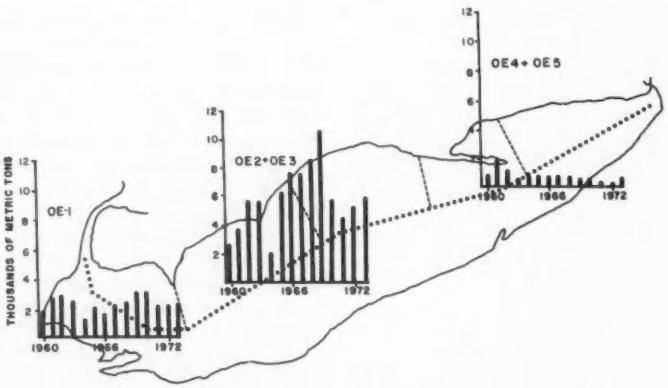

FIG. 5. Commercial production of yellow perch from Canadian waters of Lake Erie by statistical districts, 1960–73.

personal communication) while CUE has declined in all Statistical Districts except in certain portions of Ontario's waters where an increase was evident in 1971 and 1972. Effective effort has decreased in most U.S. waters.

Three- and 4-yr-old fish contributed significantly to the fishery in the 1960s (Fig. 6). However, analysis of recent Canadian catches shows a dominance by 2- and 3-yr-old fish. This trend towards smaller and younger fish is evidence that fewer perch are surviving to maturity. The effect of such exploitation can be detrimental to the resource by reducing potential egg deposition and recruitment.

Variability in year-class strength has been a characteristic of perch populations in western Lake Erie since the 1950s. Strong year-classes were recruited in 1959, 1962, 1965, and on a lesser scale in 1970 (Fig. 6 and 7). Coincidentally, strong year-classes of walleye were produced during the same years. This suggests that environmental factors, particularly water temperature during the spawning and incubation periods of walleye and perhaps of yellow perch which spawn shortly after the walleye, may play a more important role than any biological factor in determining year-class success or failure (Hartman 1973).

Since 1970, year-classes of yellow perch have not been strong. Reduced production in 1970 reflected the depletion of the very strong year-class hatched in 1965 (Fig. 6). Catches in 1972 and 1973 were dependent on the fair year-classes of 1970 and 1971, respectively.

Growth rates of yellow perch from Canadian waters of western Lake Erie have shown no significant variation from year to year since the 1950s (S. J. Nepszy unpublished data). There are slight but not significant differences between growth rates from each of the basins. Growth of yellow perch from the Central basin has generally been greater than that of perch from the Western basin. Yellow perch in the Ontario waters of the Eastern basin grow slower than those from the Western basin while in the U.S. waters of the Eastern basin they appear to grow at a rate similar to those from the Central basin (Great Lakes Fishery Commission personal communication).

This percid has now become the dominant species in the commercial harvest of Lake Erie. Although total lake landings of yellow perch have increased slightly since 1971, this does not necessarily indicate increased availability of stocks. The population continues to exhibit signs of stress and deterioration. It is experiencing poor recruitment coincident with increasing substantial exploitation of young fish. This, coupled with decreased catch per unit effort, lead us to be very concerned over the present and future status of this fishery.

WALLEYE

The discrete stocks of walleye in the Western and Eastern basins have been important to the fishery for a number of years. The stability of the smaller population in the Eastern basin has been shown by the continued presence of a greater percentage of adults from much older age-groups than that found in walleye stocks in the Western basin (Wolfert 1972). Low fishing intensity, protective management regulations by the State of New York, and the absence of the effects of accelerated eutrophication in this basin have contributed to stability in the popula-

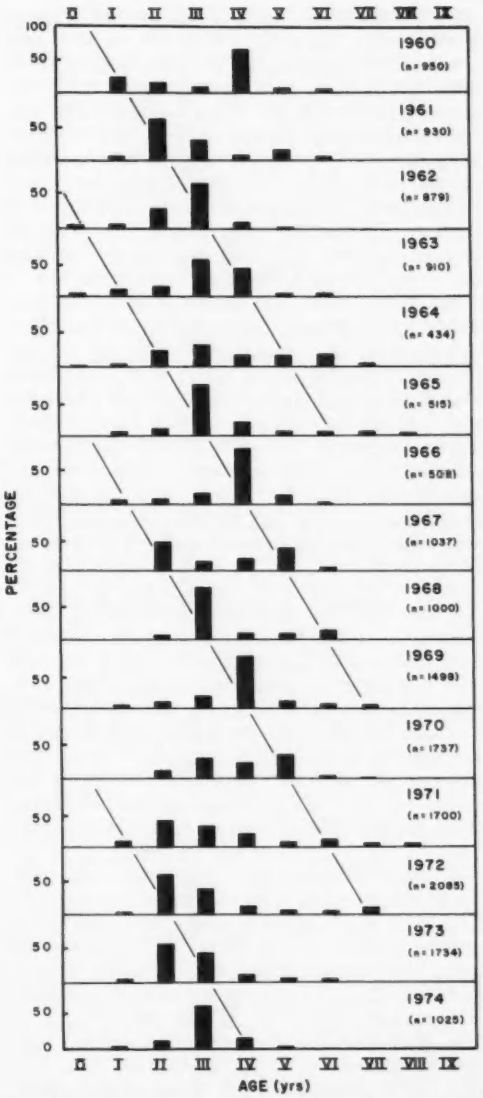

FIG. 6. Age-percentage composition of Lake Erie yellow perch from all Canadian commercial catch samples, 1960–74, all basins combined. n refers to sample size. Diagonal lines follow the strong year-classes of 1959, 1962, 1965, and the fair year-class of 1970 through the fishery. These data also generally apply to United States waters.

tion. However, recent sampling by the U.S. Fish and Wildlife Service showed few walleyes from the 1970 and 1972 year-classes (D. Wolfert personal communication). This is in direct contrast

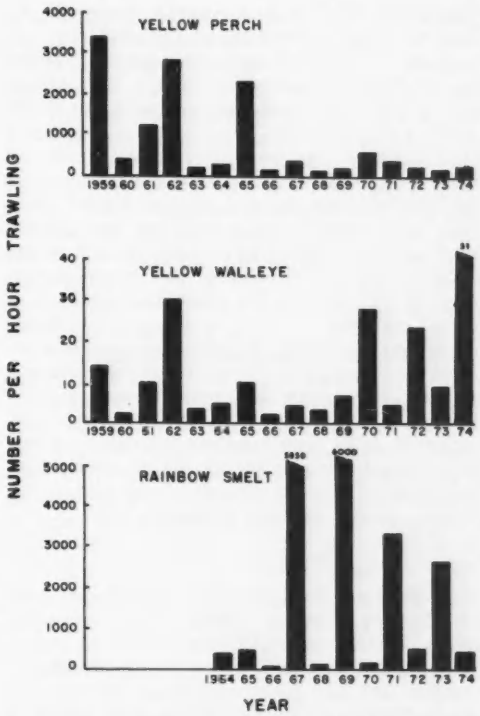

FIG. 7. Average numbers of young-of-the-year yellow perch, walleye and rainbow smelt caught per hour of trawling in Lake Erie. Yellow perch and walleye indices are from index trawling in Western basin (courtesy of U.S. Fish and Wildlife Service, Great Lakes Fishery Laboratory, Ann Arbor, Mich.) Smelt indices are from Ontario Ministry of Natural Resources index trawling in the Eastern and Central basins.

with the Western basin stock where strong cohorts were produced in those years and leads to concern over possible weakening of the historical series of regular year-classes of comparable strength.

The Western basin and western portion of the Central basin have remained closed to commercial fishing for walleye since 1970 when Ontario imposed a complete ban because of unacceptable concentrations of mercury in the flesh of the species. At the same time the state of Ohio placed a 5-yr moratorium on commercial fishing for walleye. There is no commercial fishing for walleye in Michigan waters of Lake Erie.

However, although commercial fishing for this species has been prohibited, incidental catches in other fisheries and exploitation by anglers have been allowed. Recent preliminary surveys of these

fisheries by state and provincial agencies indicate that the annual harvest is considerable.

Sampling of the present population of walleye in the Western basin shows that it is comprised mainly of three relatively strong year-classes, 1970, 1972, and 1974 (Fig. 7). It has been estimated that a large percentage of the females of the strong 1970 year-class should have spawned in 1973. Adverse environmental conditions at the time of spawning and during egg development may have contributed significantly to the failure of that year-class. However, the continued presence of the strong 1970 year-class, remnants of other year-classes, and suitable environmental conditions produced a succesful year-class in 1974. The recent ban on commercial exploitation of walleye from this area and the appearance of strong year-classes in 1970, 1972, and most recently in 1974 may improve the status of this stock. Nonetheless, management to enhance recruitment of younger year-classes into the adult stock on a sustained basis is essential.

Other Native Species

Information on other warmwater species such as white bass, channel catfish (*Ictalurus punctatus*), gizzard shad (*Dorosoma cepedianum*), and various cyprinids is scanty. However, relative indices of abundance obtained by various agencies surveying the fish resources of Lake Erie indicate that the populations of these species have remained stable for the last decade. Recent surveys in the Western and Central basins have shown that year-classes of forage fishes such as emerald shiner (*Notropis atherinoides*), spottail shiner (*Notropis hudsonius*) and trout-perch (*Percopsis omiscomaycus*) have been stable.

Freshwater drum is the most abundant species in western Lake Erie and appears to be significantly favored by the changing environment and relative decline in competitors and predators. Exploitation of this resource is incidental because of its low commercial value and lack of appeal to anglers. Recent sampling of the commercial catch indicates a population with good representation from a number of mature age-classes. Year-classes have been consistently strong in the Western basin and the prognosis is for a continued vigorous population.

Colonizing Species

First reported in Lake Erie in 1935 (Van Oosten 1937), rainbow smelt are believed to have entered via the Detroit River from an earlier introduction into the Lake Michigan watershed. Spawning runs of smelt were reported on the beaches of Point Pelee in the Western basin during the early 1940s (R. Krause personal communication). It was 1945 before smelt were first noticed from pound net catches in the western portion of the Central basin (Kennedy 1961). Subsequent records from Kennedy (1961) suggested the presence of a sizeable population by the early 1950s. With the introduction of the technique of trawling and the availability of a market for this product, fishing intensity increased markedly in 1960 and annual landings of 5,000–7,000 metric tons were common.

Rainbow smelt are almost exclusively harvested by Canadian trawlers fishing in the colder hypolimnetic and mesolimnetic waters of the Eastern Basin during the spring, fall, and winter (MacCallum and Regier 1970). Adult smelt are also available to fishermen in the Central basin during the winter and early spring. Commercial production has been relatively stable since the early 1960s (Fig. 8). This is due in part to availability of stock but perhaps is more related to consistent market demand and development of adequate storage and processing capability to meet it.

Since 1968, an alternate-year cycle of dominance is evident from the statistics obtained from commercial catch sampling (Fig. 8) and from young-of-the-year assessment (Fig. 7). Large landings have occurred when marketable 2 yr olds made up the greatest proportion of the available resource. In the alternate years, the larger 3 yr olds were available mainly in the spring but not in great abundance. Yearlings subsequently became available in the summer and fall.

If the trend continues, the 1973 year-class should become available in larger numbers in 1975 with some minor support from the 1972 year-class. However, without the presence of strong year-classes since 1971, including a relatively poor hatch in 1974, production is expected to be reduced.

A partial explanation for the alternate-year cycle of dominance is put forward by Regier et al. (1969). Although adult smelt migrate eastward across the Central basin after spawning, yearlings and young-of-the-year smelt are found near the thermocline during the summer (MacCallum and Regier 1970). Therefore, with this close association, yearlings might feed on the young-of-the-year, thereby causing an alternate-year cycle of abundance (assuming that spawning conditions are always favorable). Although limited evidence shows that smelt consume their young to a small extent (Gordon 1961; Price 1963), stomach content analyses of yearling and

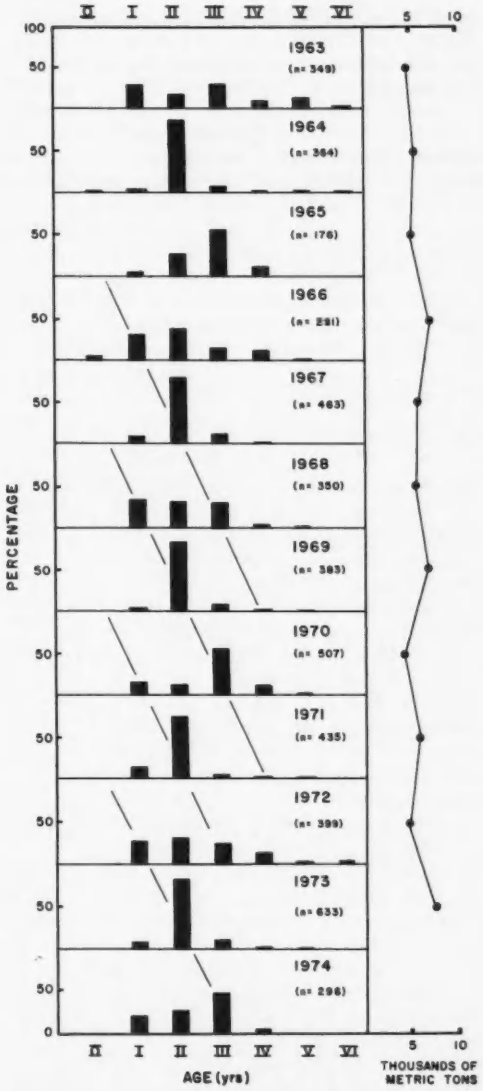

FIG. 8. Age-percentage composition of Lake Erie rainbow smelt from commercial catch samples, 1963–74. Diagonal lines follow the strong year-classes produced in alternate years. Total Canadian smelt production is shown for respective years. n refers to sample size.

older smelt associated with young-of-the-year smelt in the Eastern basin have not fully substantiated this theory of cannibalism in Lake Erie (S. J. Nepszy unpublished data).

The microsporidan, *Glugea hertwigi*, is the most abundant parasite in Lake Erie smelt and one of the greatest concern at this time. Its incidence has increased since the cysts were first observed in 1960 (Dechtiar 1965). Percentage occurrence from 1964 to 1971 rose from 4.6 to 87.9% in Eastern basin smelt and from 11.7 to 87.2% in Central basin smelt. Although the average overall occurrence had declined to 78% in 1973 and continued to decline in 1974, the recorded mortalities associated with high incidence of *G. hertwigi* (Haley 1954; Nepszy and Dechtiar 1972; Delisle 1972) indicate the importance of this parasite as a stress acting on a large population of smelt.

The sea lamprey, which invaded Lake Erie via the Welland canal, was first reported in 1921 (Van Meter and Trautman 1970). It appeared after the decline of such major species as lake trout, lake sturgeon, lake whitefish, and herring and was neither abundant nor considered a serious predator. A dearth of suitable tributary streams for spawning and nursery areas has limited the proliferation of this species in Lake Erie. Even if the fish community were restructured to include a variety and abundance of salmonids, which are the preferred prey of sea lamprey in other Great Lakes (Smith 1972), this invader would not likely become a serious problem. Recent surveys indicate that sea lamprey populations in infected Lake Erie tributaries appear to be small and restricted in distribution (Great Lakes Fishery Commission personal communication).

Salmonids have been introduced into Lake Erie since 1870. Introductions by U.S. agencies of large numbers of coho (*Oncorhynchus kisutch*) since 1968 and chinook (*O. tshawytscha*) since 1971 have been stimulated largely by increased demands of anglers. It is known that coho feed extensively on rainbow smelt and emerald shiners but what effect this predator has on the system is uncertain. There may also be some competition with other predators such as large rainbow smelt and walleye.

Investigations in 1970 showed that coho salmon migrated around the lake basin in a clockwise direction. In early spring, large numbers of coho were in the Western basin and western part of the Central basin and during the summer months moved along the north shore. By autumn, most migrating adults returned to their planting sites on the south shore but some attempted to spawn in northshore streams in the Eastern basin (R. Scholl personal communication).

Alewives, first reported in Lake Erie in 1931 (Dymond 1932), have never become abundant

TABLE 2. Summary of cultural stresses affecting commercial fish species in Lake Erie (X, moderate effect; XX, severe effect).

Species	Stresses					
	Commercial fishery	Nutrient loading	Nonnative species	Erosion and siltation	Tributary and shoreline restructuring	Toxic materials
Lake sturgeon	X			X	X	
Lake trout	XX	X	X(?)	X		
Lake whitefish	X	X		X		
Lake herring	XX	X		X		
Sauger	XX	X		X	X	
Blue pike	XX	X	X			
Walleye	XX	X	X	X	X	X(?)
Yellow perch	XX		X			

to the extent that they have in Lake Michigan and Lake Ontario. Apparent lack of success can be attributed to the abundance of predators (Smith 1970) and low temperature regimes in winter and spring. According to Smith (1968), availability of deep-water regions where alewives could concentrate in the winter to avoid cold water is limited in Lake Erie.

Although the 1973 and 1974 year-classes of alewife have been considered strong by state and provincial agencies, recently reported mortalities of young-of-the-year and yearlings, especially in the Eastern basin, may prevent the anticipated utilization of this resource as a strong forage base.

Discussion

The major stresses acting on the fish community may be defined as the commercial fishery and reduced water quality, including nutrient loading. Other influences such as the introduction of nonnative species, erosion and siltation, tributary and shoreline restructuring, and the introduction of toxic materials have been significant factors affecting some species (Table 2). We acknowledge, however, that the effects of some of these stresses have not been fully evaluated. An increase in mean annual temperature resulting from a general climatic warming in the Great Lakes area may have been disadvantageous to lake whitefish and lake herring. This stress was, of course, beyond the influence of man.

The dominant and most consistent cultural stress acting on the community has been the commercial fishery. All species important to man have been affected. Throughout its history the commercial fishery of Lake Erie has been inadequately regulated (Regier et al. 1969). The progressive exploitation of stocks in a multi-species fishery has been described by Regier and Loftus (1972) as the "fishing-up" sequence. Basically, this process is described as the gradual shift in fishing effort from higher-valued to lower-valued species as the preferred stocks pass their peaks and decline in abundance. It encompasses the ability of fishermen to increase fishing efficiencies through improved gear, more and larger vessels and a growing knowledge of the behavior of fish. This process began in Lake Erie in the 1800s and is still continuing.

Degradation of environment as a result of nutrient loading has placed additional stresses on most commercial species. Hypolimnetic oxygen depletion in the Central basin in summer causes certain species to move more than usual or to concentrate in the deeper water of the Eastern basin and, therefore, be more vulnerable to exploitation by the fishery. Regier and Loftus (1972) point out that sound resource management on a long-term basis would require a reduction in exploitation pressure where cultural eutrophication is causing an increase in efficiency of capture.

There appears to be a relation between eutrophication and incidence of parasitism in Lake Erie. Dechtiar (1972) recorded 215 species of parasites from 46 species of fish collected between 1961 and 1969. Ninety-six percent of the 1,112 fish examined were infected by at least one species of parasite. This represents a major increase over the 112 species recorded by Bangham and Hunter (1939) from 78 species of fish. Dechtiar considered that mortality among yearling yellow perch and freshwater drum in 1963 may have been due to a heavy infestation of *Ichthyophthyrius multifiliis*. Nepszy and Dechtiar (1972) suggested that an unusually severe post-spawning mortality of smelt in 1971 may have been due partly to the microsporidan parasite, *Glugea hertwigi*, which infested 97.3% and 76.6%

of dead males and females, respectively. Dechtiar (1972) considers that the increase in numbers and variety of parasites is probably due to changes in the environment and fish fauna. Snieszko (1974) believes that the incidence of infectious diseases in fish is related to stress caused by eutrophication, domestic and industrial pollution, pesticides, and temperature but states that the processes involved in these relations are not well understood.

Smith (1969) and Christie (1974) suggested a further general effect of eutrophication which could possibly lead to change in species composition. Species such as yellow perch, smelt, freshwater drum, fathead minnow (*Pimephales promelas*), spottail shiner, and some of the centrarchids are able to tolerate considerable levels of pollution, particularly during the critical egg development stage. Eutrophication could be favoring survival of these species and may be a partial explanation for the recent dominance of rainbow smelt and yellow perch in Lake Erie.

Of the three invading species usually regarded as nuisances in the Great Lakes, sea lamprey and alewife have failed to establish major populations in Lake Erie while rainbow smelt has been and continues to be exploited successfully by the commercial fishery. However, rainbow smelt may have affected the blue pike and walleye through predation of their young and competition (Regier et al. 1969). The full extent of this interspecific action is not known.

Currently, the physiological effects of toxic materials on fish are not well understood and, therefore, the most obvious effects resulting from the introduction of these chemicals to the lake appear to be socioeconomic. For example, the discovery of mercury in fish at concentrations greater than 0.5 ppm led to the suspension of fishing for walleye and white bass in 1970. The ban has been partially lifted for white bass, and some walleye are also taken in that fishery, but the overall effect has been a substantial loss of income to the commercial fishery. However, the ban appears to have affected Western basin walleyes directly by allowing the stocks to recover in the absence of serious exploitation.

Regier (1973) considers that exploitation, eutrophication, and the introduction of nonnative species, acting either individually or synergistically, have altered the structure of Lake Erie's fish community from dominance by large benthic forms to dominance by small pelagic forms. With this change in the fish community there has been an increase in year-to-year fluctuations in the harvest of certain species and a lower market value per unit of biomass. This has resulted in a gradual transition so that the commercial fishery is now more capital intensive and more efficient in harvesting fish biomass, thus maintaining a reasonably constant return on investment and labor (Lambert 1975). However, because of limited market demand, low-valued, coarse species such as freshwater drum, carp (*Cyprinus carpio*), and gizzard shad are not now purposely exploited in large quantities. New incentives in product and market development could increase consumer demand for these species. However, the further broadening of the fishery to exploit underutilized species would add to fishery management problems such as estimation of total sustainable yields (Edwards and Hennemuth 1975). Present studies on the biology of high-valued species must be increasingly supported by studies on other species in the community.

Christie (1974) points out that in the Great Lakes there is no economic advantage to the commercial fishery to harvest fish stocks at lower trophic levels and makes a strong plea for the restoration of premium fishes wherever possible. On the other hand, Hubschman (1971) considers that energy production in Lake Erie is in excess of that which can be efficiently harvested by the traditional fishery. He calls for the introduction of harvestable macroinvertebrates to utilize the abundance of primary organic matter and detritus which has resulted from eutrophication. We favor Christie's preference for high-valued fish but recognize that Lake Erie has limitations which could preclude the rehabilitation of salmonids. The location of Lake Erie at the southern limit of North American distribution of lake trout, lake whitefish, and lake herring makes restoration of these species doubtful in view of the additional stresses referred to above. We reject Hubschman's suggestion that omnivorous macroinvertebrates should be introduced into Lake Erie. In our view the lake is well-suited for the production of mesothermic percids (walleye and perch) and we feel that research and management efforts should be directed towards the building-up and maintenance of these stocks. Environmental requirements of percids are not nearly as stringent as those for salmonids; hence there is more likelihood that water quality criteria for the long-term success of walleye and perch can be attained. Smith (1972) is pessimistic about the future of percids in lakes St. Clair, Erie, and Ontario because of degradation of water quality. We share the more optimistic views of Regier et al. (1969) and Christie (1974) that the joint efforts of the U.S. and Canada will lead to improvement in water quality in the near future.

We consider that major concern for the im-

mediate future of the Lake Erie fishery is regulation of exploitation. Recent strong year-classes and reduced commercial exploitation on the western Lake Erie walleye may lead to an improvement in the status of this stock, and restrictions on the future harvest are required if the fishery is to survive. The present status of the yellow perch population should be sufficient evidence of the dangers of overexploitation. The implementation of careful management is urgent.

Summary and Conclusions

The fish community in Lake Erie is a mixture of coldwater, mesothermal, and warmwater species with a predominance of warmwater and mesothermal types. Twenty-four families of fishes are represented by 114 species. Cyprinids, percids, catostomids, centrarchids, and ictalurids contribute 74% of the species. Coldwater species are now few and are largely restricted to the Eastern basin where the water is deeper and colder in summer.

Lake Erie, because of its southern location, shallowness, and fertile drainage basin, is the most productive of the Great Lakes. It has almost always led Great Lakes fisheries in variety of fish species and abundance. Nineteen species have been significant in commercial production which has averaged about 23,000 metric tons annually since 1915. Although production has remained fairly consistent, species composition has changed dramatically during the past 150 yr.

The Lake Erie ecosystem is a product of its history, which, since the arrival of European man to the drainage basin, has been a record of cultural perturbations. The conversion of forests and savannas to farmland led to increased turbidity and siltation in tributaries and inshore areas of the lake. Construction of dams and restructuring of shorelines and shallow bays further reduced spawning and nursery areas for some species. Domestic and industrial pollution increased directly with population. The increased inflow of nutrients led to excessive primary production which together with organic matter added as sewage has caused increased turbidity and widespread hypolimnetic oxygen depletion. This, in turn, has altered benthic communities with the virtual elimination of mayfly nymphs and an increase in numbers of pollution-tolerant oligochaetes. Biomagnification of toxic materials has affected the usability of certain fish species and perhaps depleted their populations as well.

Overriding the degradation of the aquatic environment has been an intensive commercial fishery. Since it began in 1795 the fishery has experienced changes in gear and technology which have greatly increased efficiency. Before 1954, U.S. production exceeded that of Canada. The Canadian fishing fleet now includes many large vessels equipped for gillnetting and trawling.

The cultural stresses listed above, in addition to the introduction of nonnative species, are considered to be responsible for changes in species composition. The highly valued lake trout, lake herring, lake whitefish, sauger, and blue pike have virtually disappeared. Commercial production of medium-priced species such as yellow perch and white bass has increased and lower-valued species such as rainbow smelt and freshwater drum have become more prevalent in the fishery.

Yellow perch is now the principal species in the commercial fishery. Fishing pressure on yellow perch increased in the 1950s as the harvest of whitefish and blue pike declined. The recent ban on walleye because of mercury contamination shifted further effort to the harvest of yellow perch. The population is experiencing considerable variability in year-class strength and the trend is towards younger fish in commercial landings. Poor recruitment in the face of increased exploitation of young fish indicates that the future of this species is uncertain.

With the presence of recent strong year-classes and the ban on commercial fishing for this species by the Province of Ontario and the State of Ohio since 1970, the walleye stocks of the Western basin have shown signs of improvement. The maintenance of a strong adult population will depend to a large extent on what restrictions on exploitation are adopted when commercial fishing is resumed.

Rainbow smelt, which is harvested mainly by Canadian trawlers, is the second most important species in the fishery. Production has been relatively stable since the early 1960s due to availability of stock and market demand. The incidence of a microsporidan parasite in Lake Erie smelt is high. Although there is some evidence that the parasite may be associated with mortality, long-term effects on the population have yet to be determined.

Because of limited market demand, low-valued species such as freshwater drum, carp, and gizzard shad are not purposely exploited in quantity. These species appear to be favored by the changing environment and lack of competition.

The future of the Lake Erie fishery depends primarily on what efforts are expended to improve water quality and to regulate exploitation. The joint efforts of the U.S. and Canada to control pollution are expected to improve the environment. Restoration to levels conducive to salmonid production may be unrealistic but it is

likely that water quality criteria for the long-term success of percids can be attained. Clearly, the future of these valuable stocks will depend on how they are managed.

Acknowledgments

We gratefully acknowledge the constructive comments of A. H. Berst, N. M. Burns, A. H. Lawrie, N. V. Martin, A. M. McCombie, H. A. Regier, J. D. Roseborough, and R. L. Thomas who critically reviewed the manuscript.

APPLEGATE, V. C., AND H. D. VAN METER. 1970. A brief history of commercial fishing in Lake Erie. U.S. Fish Wildl. Serv. Fish. Leafl. 630: 28 p.

BALDWIN, N. S., AND R. W. SAALFELD. 1962 (plus supplement 1970). Commercial fish production in the Great Lakes, 1867–1960 [supplement 1961–68]. Great Lakes Fish. Comm. Tech. Rep. 3: 166 p.

BANGHAM, R. V., AND G. W. HUNTER. 1939. Studies on fish parasites of Lake Erie: distribution studies. Zoologica (N.Y.) 24: 385–448.

BEETON, A. M. 1966. Indices of Great Lakes eutrophication. Great Lakes Res. Div. Univ. Mich. Publ. 15: 1–8.

1969. Changes in the environment and biota of the Great Lakes, p. 150–187. *In* Eutrophication: causes, consequences, correctives. Natl. Acad. Sci. Washington, D.C.

BEETON, A. M., AND W. T. EDMONDSON. 1972. The eutrophication problem. J. Fish. Res. Board Can. 29: 673–682.

BRINKHURST, R. O., A. L. HAMILTON, AND H. B. HERRINGTON. 1968. Components of the bottom fauna of the St. Lawrence Great Lakes. Univ. Toronto, Great Lakes Inst. 33: 50 p.

BRITT, N. W. 1955. Stratification in western Lake Erie in summer of 1953: effects on the *Hexagenia* (Ephemeroptera) population. Ecology 36: 239–244.

BURNS, N. M. 1976. Temperature, oxygen, nutrient distribution patterns in Lake Erie, 1970. J. Fish. Res. Board Can. 33: 485–511.

CARR, J. F. 1962. Dissolved oxygen in Lake Erie, past and present. Great Lakes Res. Div. Univ. Mich. Publ. 9: 1–14.

CARR, J. F., V. C. APPLEGATE, AND M. KELLER. 1965. A recent occurrence of thermal stratification and low dissolved oxygen in western Lake Erie. Ohio J. Sci. 65: 319–327.

CARR, J. F., AND J. K. HILTUNEN. 1965. Changes in the bottom fauna of western Lake Erie from 1930 to 1961. Limnol. Oceanogr. 10: 551–569.

CHRISTIE, W. J. 1974. Changes in the fish species composition of the Great Lakes. J. Fish. Res. Board Can. 31: 827–854.

DECHTIAR, A. O. 1965. Preliminary observations on *Glugea hertwigi* Weissenberg, 1911 (Microsporidia: Glugeidae) in American smelt *Osmerus mordax* (Mitchill) from Lake Erie. Can. Fish. Cult. 34: 35–38.

1972. New parasite records for Lake Erie fish. Great Lakes Fish Comm. Tech. Rep. 17: 20 p.

DELISLE, C. 1972. Variations mensuelles de *Glugea hertwigi* (Sporozoa: Microsporida) chez différents tissus et organes de l'eperlan adulte dulcicole et conséquences de cette infection sur une mortalité massive annuelle de ce poisson. Can. J. Zool. 50: 1589–1600.

DOBSON, H. H., AND M. GILBERTSON. 1972. Oxygen depletion in the hypolimnion of the central basin of Lake Erie, 1929 to 1970, p. 3–8. *In* N. M. Burns and C. Ross [ed.] Project Hypo: an intensive study of the Lake Erie central basin hypolimnion and related surface water phenomena. Can. Cent. Inland Waters Pap. No. 6.

DYMOND, J. R. 1932. Records of the alewife and steelhead (rainbow trout) from Lake Erie. Copeia 1: 32–33.

EDWARDS, R., AND R. HENNEMUTH. 1975. Maximum yield: assessment and attainment. Oceanus 18: 3–9.

FERGUSON, R. G., AND A. J. DERKSEN. 1971. Migrations of adult and juvenile walleyes (*Stizostedion vitreum vitreum*) in southern Lake Huron, Lake St. Clair, Lake Erie and connecting waters. J. Fish. Res. Board Can. 28: 1133–1142.

FRY, F. E. J. 1969. Some possible physiological stresses induced by eutrophication, p. 531–536. *In* Eutrophication: causes, consequences, correctives. Natl. Acad. Sci. Washington, D.C.

GORDON, W. G. 1961. Food of the American smelt in Saginaw Bay, Lake Huron. Trans. Am. Fish. Soc. 90: 439–443.

HALEY, A. J. 1954. Microsporidian parasite, *Glugea hertwigi* in American smelt from the Great Bay region, New Hampshire. Trans. Am. Fish. Soc. 83: 84–90.

HARTMAN, W. L. 1972. Lake Erie: effects of exploitation, environmental changes, and new species on the fishery resources. J. Fish. Res. Board Can. 29: 899–912.

1973. Effects of exploitation, environmental changes, and new species on the fish habitats and resources of Lake Erie. Great Lakes Fish. Comm. Tech. Rep. 22: 43 p.

HOLLING, C. S. 1973. Resilience and stability of ecological systems. Annu. Rev. Ecol. Syst. 4: 1–23.

HUBSCHMAN, J. H. 1971. Lake Erie: pollution abatement, then what? Science 171: 536–540.

INTERNATIONAL JOINT COMMISSION. 1969. Report of the International Joint Commission, United States and Canada on the pollution of Lake Erie, Lake Ontario and the international section of the St. Lawrence River. Lake Erie. 2: 316 p.

KEMP, A. L. W., T. W. ANDERSON, R. L. THOMAS, AND A. MUDROCHOVA. 1974. Sedimentation rates and recent sediment history of Lakes Ontario, Erie and Huron. J. Sediment. Petrol. 44: 207–218.

KENNEDY, W. A. 1961. Daily catch records of Crewe Brothers Fishery, Lake Erie — 1904 to 1956. Fish. Res. Board Can. MS Rep. Ser. (Biol.) 706: 411 p.

LAMBERT, L. 1975. Ontario's Lake Erie commercial fishery — a social and economic profile. Commer. Fish Fur Branch, Ont. Minist. Nat. Resour. 38 p.

LANGLOIS, T. H. 1945. Water, fishes, and cropland management. Trans. North Am. Wildl. Conf. 10: 190–196.

LAWLER, G. H. 1965. Fluctuations in the success of year classes of whitefish populations with special reference to Lake Erie. J. Fish. Res. Board Can. 22: 1197–1227.

MACCALLUM, W. R., AND H. A. REGIER. 1970. Distribution of smelt, *Osmerus mordax*, and the smelt fishery in Lake Erie in the early 1960's. J. Fish. Res. Board Can. 27: 1823–1846.

McALLISTER, D. E. 1970. Rare or endangered Canadian fishes. Can. Field Nat. 84: 5–8.

MOENIG, J., N. MACLEAN, AND J. SCHWEIGERT. 1974. The lake trout of Lake Erie: an historical review. Univ. Toronto, Ont. 50 p.

NEPSZY, S. J., AND A. O. DECHTIAR. 1972. Occurrence of *Glugea hertwigi* in Lake Erie rainbow smelt (*Osmerus mordax*) and associated mortality of adult smelt. J. Fish. Res. Board Can. 29: 1639–1641.

PARSONS, J. W. 1967. Contributions of year-classes of blue pike to the commercial fishery of Lake Erie, 1943–59. J. Fish. Res. Board Can. 24: 1035–1066.

1970. Walleye fishery of Lake Erie in 1943–62 with emphasis on contributions of the 1942–61 year-classes. J. Fish. Res. Board Can. 27: 1475–1489.

PRICE, J. W. 1963. A study of the food habits of some Lake Erie fish. Bull. Ohio Biol. Surv. (New Ser.) 2: 1–89.

REGIER, H. A. 1973. Sequence of exploitation of stocks in multispecies fisheries in the Laurentian Great Lakes. J. Fish. Res. Board Can. 30: 1992–1999.

REGIER, H. A., V. C. APPLEGATE, AND R. A. RYDER. 1969. The ecology and management of the walleye in western Lake Erie. Great Lakes Fish. Comm. Tech. Rep. 15: 101 p.

REGIER, H. A., AND W. L. HARTMAN. 1973. Lake Erie's fish community: 150 years of cultural stresses. Science 180: 1248–1255.

REGIER, H. A., AND K. H. LOFTUS. 1972. Effects of fisheries exploitation on salmonid communities in oligotrophic lakes. J. Fish. Res. Board Can. 29: 959–968.

SCOTT, W. B. 1951. Fluctuations in abundance of the Lake Erie cisco (*Leucichthys artedi*) populations. Contrib. R. Ont. Mus. Zool. 32: 41 p.

SCOTT, W. B., AND E. J. CROSSMAN. 1973. Freshwater fishes of Canada. Bull. Fish. Res. Board Can. 184: 966 p.

SLY, P. G. 1976. Lake Erie and its basin. J. Fish. Res. Board Can. 33: 355–370.

SMITH, L. L. 1969. Effect of eutrophication on fish and related organisms. Minn. Univ. Water Resour. Res. Cent. Bull. 13: 15–18.

SMITH, S. H. 1968. Species succession and fishery exploitation in the Great Lakes. J. Fish. Res. Board Can. 25: 667–693.

1970. Species interactions of the alewife in the Great Lakes. Trans. Am. Fish. Soc. 99: 754–765.

1972. The future of salmonid communities in the Laurentian Great Lakes. J. Fish. Res. Board Can. 29: 951–957.

SNIESZKO, S. F. 1974. The effects of environmental stress on outbreaks of infectious diseases of fishes. J. Fish Biol. 6: 197–208.

TRAUTMAN, M. B. 1957. The fishes of Ohio. Ohio State Univ. Press, Columbus, Ohio. 683 p.

VAN METER, H. D., AND M. B. TRAUTMAN. 1970. An annotated list of the fishes of Lake Erie and its tributary waters exclusive of the Detroit River. Ohio J. Sci. 70: 65–78.

VAN OOSTEN, J. 1937. First records of the smelt, *Osmerus mordax*, in Lake Erie. Copeia. 1: 64.

WATT, K. E. F. 1973. Principles of environmental science. McGraw-Hill, Inc. Toronto, Ont. 319 p.

WOLFERT, D. R. 1963. The movement of walleyes tagged as yearlings in Lake Erie. Trans. Am. Fish. Soc. 92: 414–420.

1969. Maturity and fecundity of walleyes from the eastern and western basins of Lake Erie. J. Fish. Res. Board Can. 26: 1877–1888.

1972. Age and growth of Lake Erie walleye. U.S. Bur. Sport Fish. Wildl., Sandusky, Ohio. 34 p.

Processes Within Lake Erie

N. M. Burns, J.-M. Jaquet, A. L. W. Kemp, D. C. L. Lam, J. H. Leach, M. Munawar, T. J. Simons, P. G. Sly, R. L. Thomas, N. H. F. Watson, and J. D. H. Williams[1]

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Burns, N. M., J.-M. Jaquet, A. L. W. Kemp, D. C. L. Lam, J. H. Leach, M. Munawar, T. J. Simons, P. G. Sly, R. L. Thomas, N. H. F. Watson, and J. D. H. Williams. 1976. Processes within Lake Erie. J. Fish. Res. Board Can. 33: 639–643.

The current level of knowledge concerning Lake Erie is inadequate for the development of a predictive model of the system. Nevertheless, an outline of the submodels required for the development of a predictive model is presented. Further, as a result of many recent studies, the level of understanding of the Lake Erie system has increased immensely and major gaps in the present knowledge of the lake have been identified; useful areas for future research have been indicated. Many of the recent studies on Lake Erie are included in this special issue. The results of these investigations have been summarized and linked together where possible, to describe the lake and give an overview of the more important processes in Lake Erie.

Burns, N. M., J.-M. Jaquet, A. L. W. Kemp, D. C. L. Lam, J. H. Leach, M. Munawar, T. J. Simons, P. G. Sly, R. L. Thomas, N. H. F. Watson, and J. D. H. Williams. 1976. Processes within Lake Erie. J. Fish. Res. Board Can. 33: 639–643.

L'état actuel de nos connaissances du lac Erié ne nous permet pas encore de développer un modèle prophétique du système. Néanmoins, nous donnons une esquisse des sous-modèles requis pour le développement d'un tel modèle. De plus, plusieurs études récentes ont considérablement rehaussé notre niveau de compréhension du système du lac Erié et permis d'identifier les lacunes majeures dans nos connaissances actuelles du lac; des domaines prometteurs de recherches futures ont été précisés. Ce numéro spécial contient plusieurs des récentes études sur le lac Erié. Nous résumons les résultats de ces recherches et, autant que possible, en faisons la synthèse dans le but de décrire le lac et de donner une vue d'ensemble des plus importants processus en jeu dans le lac Erié.

Accepted November 18, 1975 Accepté le 18 novembre 1975

Synthesis of Processes into an Ecological Model

At the start of the project to publish "Lake Erie in the Early Seventies," the contributors to this special issue hoped that the various studies described here could be linked together in an overall quantitative model of Lake Erie which would have short-term predictive capabilities. Failing this, it was anticipated that at least a qualitative description of processes in the lake could be given such that water movements, sediment characteristics, and nutrient loadings could be linked to explain phytoplankton, zooplankton, and fish distributions.

After the first few discussions of the Lake Erie Group, it was realized that neither of these two aims could be fully attained even though the data assemblage was much greater than that usually available for building a detailed simulation model. (A simulation model being considered here as a model in the developmental stage whereby predictions of values of variables by the model are made for the purpose of comparison with measured values.) The Group discussions revealed that in nearly every study, some necessary rate process had not been investigated with the consequence that the results and findings of the study remained inadequate for incorporation into a truly predictive model. A conceptual outline of a complete model for the Lake Erie System is shown in Fig. 1. A predictive model is defined here as a model that is considered to be in the final stages of development such that predictions of future situations by the model will have known limits of confidence and thus can be used in the formulation of management plans. The state of the art (as shown in Fig. 1) is such that only one model formulated for Lake Erie is considered to have

[1]These authors formed the Lake Erie Group, referred to throughout this paper.

Printed in Canada (J4072)
Imprimé au Canada (J4072)

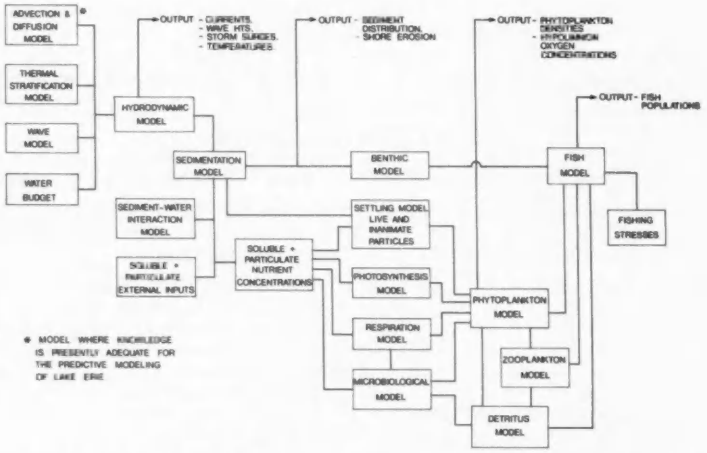
ADVECTION & DIFFUSION MODEL
THERMAL STRATIFICATION MODEL
WAVE MODEL
WATER BUDGET
HYDRODYNAMIC MODEL
OUTPUT - CURRENTS, - WAVE HTS. - STORM SURGES. - TEMPERATURES.
OUTPUT - SEDIMENT DISTRIBUTION, - SHORE EROSION
OUTPUT - PHYTOPLANKTON DENSITIES - HYPOLIMNION OXYGEN CONCENTRATIONS
OUTPUT - FISH POPULATIONS
SEDIMENTATION MODEL
BENTHIC MODEL
FISH MODEL
SEDIMENT-WATER INTERACTION MODEL
SETTLING MODEL LIVE AND INANIMATE PARTICLES
FISHING STRESSES
SOLUBLE + PARTICULATE EXTERNAL INPUTS
SOLUBLE + PARTICULATE NUTRIENT CONCENTRATIONS
PHOTOSYNTHESIS MODEL
PHYTOPLANKTON MODEL
RESPIRATION MODEL
ZOOPLANKTON MODEL
MICROBIOLOGICAL MODEL
DETRITUS MODEL
* MODEL WHERE KNOWLEDGE IS PRESENTLY ADEQUATE FOR THE PREDICTIVE MODELING OF LAKE ERIE

FIG. 1. Schematic outline of the more important data inputs, models, and linkages between models required to formulate a predictive model for a whole-lake system. (Many possible linkages have been omitted to maintain a manageable level of simplicity.)

reached the required capability for incorporation into a general predictive model.

Nevertheless, a start has been made on the general quantitative model by Lam and Jaquet (1976), who have linked the physical, chemical, and sedimentological characteristics of the lake to develop a good simulation model for total phosphorus distributions in Lake Erie during a storm period. We don't know whether this model could be extended to predict phytoplankton biomass levels in the lake because the level of biological knowledge required to develop such a predictive model remains unknown. In general, it has not been possible to link the physical, inorganic systems with the biological, organic systems to a degree that is better than a common-sense level such as: "the phytoplankton densities are higher in the Western basin than in the other basins due to the high nutrient loading to that basin." The difficulty in integrating the two major areas arises particularly from the complexity of biological systems. These systems require an immense amount of data and analysis to define their states with a certitude comparable to that presently attainable with the physical, chemical, and geological systems.

A number of processes have been identified as being inadequately investigated in the studies reported in this issue. These processes would have to be thoroughly studied if a systems model with predictive capabilities were to be developed for Lake Erie. The required studies are:

a) Careful measurements of the pattern of water mass exchange between the three basins, especially in regard to the exchange between the hypolimnia of the Central and Eastern basins;

b) An investigation of the resuspension of fine-grained sediments by currents and wind-generated waves, together with the extraction of nutrients from the sediments as a result of the hydrodynamic action;

c) An investigation of storm-episode sediment transport within the lake in contrast to sediment transport under non-storm conditions;

d) A careful examination of the sediment–water interface with emphasis on the form of nutrients and toxic materials in the centimeter below the interface and in the meter of water above;

e) River inputs of phosphorus, nitrogen, and suspended solids, in particular, should be examined as a function of flow so that accurate loading estimates of materials can be calculated; atmospheric inputs of nutrient elements should also be measured;

f) The apatite content of river waters should be measured and a test of apatite solubility in Lake Erie waters and sediment interstitial waters should be carried out;

g) The loading estimates of fine-grained materials and associated elements to the sediments should be improved by ensuring adequate basin coverage in the cores examined;

h) A major study of aerobic heterotroph bacterial activity in the lake should be carried out in conjunction with studies (i) and (j), described below;

i) Respiration and other oxygen-consuming

processes should be studied to the same extent as a complementary study of primary productivity. These processes should be investigated both in the water and at the sediment–water interface;

j) The rates of decay and formation of detritus and soluble organic material should be carefully determined;

k) Phytoplankton species and densities, together with their associated productivity at all depths in the lake, should be determined;

l) The vertical migration of both phytoplankton and zooplankton should be studied in relation to epilimnion nutrient budgets;

m) The cause of major algal blooms should be determined if possible;

n) Rotifers must be included in the study of the zooplankton and the grazing patterns of all the major species should be examined;

o) A lake-wide program to study the benthos should be carried out;

p) The effect of fish on phytoplankton and zooplankton populations should be determined.

If these studies were carried out, the general understanding of the Lake Erie System would increase immensely but, even then, it would remain doubtful whether a complete predictive model for Lake Erie, as shown in Fig. 1, could be finalized. At this stage it is probable that the overall complete model could be formulated but that a number of other processes for the development of the model would also be identified and found to require investigation. These investigations would then result in the formulation of an improved version of the overall model. It would seem, therefore, that the development of a predictive lake systems model is too complex to be achieved by the implementation of a single master-plan of investigation. It is more probable that the model would evolve as a result of first having set the achievement of the model as a goal and subsequently ensuring that a number of investigators work on the identified gaps in knowledge on a continuing basis. The work could continue indefinitely with the degree of confidence in the predictions of the model increasing with time. It would seem that environmental models will be increasingly complex until it is possible to discover the key variables and processes which will permit the formulation of simple models capable of good predictions.

Processes of Major Importance in Lake Erie

A brief outline of the major processes described in the studies that comprise this publication is presented in this final section. Unfortunately, at the time of preparation of this work, there were no lake-wide studies of the benthic fauna of Lake Erie available; therefore, a list of references which relate to the benthic fauna of the lake are appended to this article (Appendix 1).

The predominant flow in Lake Erie is from west to east as is the prevailing wind direction, with the flow of water through the Western basin being relatively independent of wind conditions over the lake. The currents in the Central basin are dependent on weather conditions but west to east nearshore currents tend to prevail at most times. The waters of the Eastern basin are relatively well mixed as they are much affected by wind.

The Western basin has an accretion of fine-grained sediments in waters of less than 10 m depth, whereas in the Eastern and Central basins, the fine-grained sediments are washed out of the areas of this depth with the deposition occurring in the deeper parts of these basins. This difference between sedimentation patterns in the basins probably relates to the fact that even under normal storm conditions, the flow of water from the Western basin continues to be the hydraulic flow resulting from the inflowing rivers, whereas the flow pattern between the Central and Eastern basins is considerably affected by wind. Also, a sediment budget condition occurs in the Western basin in which input to the basin exceeds the export to the Central basin. This implies that, during storms, turbid Western basin waters cannot be flushed out of the basin by less turbid Central basin waters. Under these conditions, very little sediment can be exported from the Western basin as the sediment resuspended during a storm mostly resettles within the Western basin after the storm. The greater degree of mixing between the waters of the Central and Eastern basins during storms does permit a much larger sediment transport from the Central basin to the Eastern basin.

Mercury in the sediments was found to be related to the predominant pathway of water flow through the lake. The areas of mercury contamination of the sediments due to chlor-alkali plant discharges can be distinguished from the non-contaminated areas by means of two different organic carbon–mercury association relations.

The sand–silt size of apatite particles loaded into Lake Erie were found to be retained in the coarser, nearshore sediments with the finer, clay-size particles being retained in the depositional basins of the lake. Phosphorus is found only in the apatite form in the coarse-grained sediments whereas it occurs in the organic phosphorus, iron-associated phosphorus, and apatite forms in the fine-grained sediments.

The apatite in the lake is of detrital origin and seems to have sedimented at a fairly constant rate over the past 100 yr. The sedimentation rates of

the other phosphorus forms appear to have been increasing with time. This deduction depends on the assumption of negligible phosphorus migration up the sediment column. It would appear that the phosphorus in the sediment column has been stabilized by the heavy loadings of clay-bound iron to the sediments and the iron to phosphorus ratio in Lake Erie sediments is found to be comparatively high. The concentration of nonapatite phosphorus in the fine-grained material of the surficial sediments is surprisingly constant over the whole lake and appears to be quite independent of proximity to either phosphorus or sediment sources.

The heavy loading of sediment to the Central and Eastern basins derives from the erosion of unconsolidated bluffs which are not underlain by bedrock at the lake water level; comparatively little sediment is being carried into these basins by rivers. Most of the material is eroded into the Central basin and then transported to the Eastern basin for final deposition. Pollen dating has made possible the establishment of sediment horizons which, in turn, has led to estimates of loading of many elements to the lake sediments. The loadings of anthropogenically derived metals in the Eastern basin are high. In general, the loadings of both organic and inorganic materials to the sediments of the Eastern basin are much greater than those to the Western and Central basin sediments.

Iron and manganese were both found to be present almost entirely in the particulate form in the waters of the lake. The manganese is released from the sediment in soluble form with subsequent conversion into the particulate form, while iron in the waters is associated with the resuspended clays and does not pass through the soluble phase under oxygenated conditions.

Loadings of suspended solids and soluble nutrients to Lake Erie from a number of northshore watersheds have been studied in detail. The comment was made that the estimation of stream loadings which are obtained from the multiplication of mean flows by mean concentrations, implies a linear relation between flow and concentration but, since this relation is not known for the northshore watersheds, the calculated loadings from them are considered to be rough estimates. Nevertheless, it was found that the ratio of soluble reactive phosphorus to total phosphorus increased with the yield of total phosphorus (tons P km^{-2}) draining from the land.

Changes in Lake Erie water quality are strongly associated with both agricultural and industrial development. It appears that major increases in loading began a few years prior to 1940 with the loadings continuing to increase since that time. These increased loadings are reflected in the increased concentrations of phorphorus, heavy metals, and pesticides found in recent sediments.

Extensive areas of anoxic water were observed in the Central basin hypolimnion during 1970 and resulted in large-scale regeneration of phosphorus from the sediments. The nutrient concentrations in the southshore area of the Central basin were higher than those in the vicinity of the northshore. The higher concentrations were a result of the heavy southshore loadings to the lake and were not due to nutrient release from the sediments resulting from wave action. It appears that these higher nutrient levels resulted in proportionately higher concentrations of organic materials in the southshore waters.

Although the Central basin generally has higher algal concentrations than the Eastern basin, the areal hypolimnion oxygen uptake rate of the Central basin was found to be half that of the Eastern basin rate. However, the volumetric hypolimnion oxygen depletion rate of the Central basin was found to be twice that of the Eastern basin.

The elimination of phosphorus from water to the sediment was found to be directly related to the total phosphorus concentration in the water and inversely related to water depth. Epilimnion nutrient budgets, together with phosphorus distribution maps, revealed that a major decrease in water-borne phosphorus loadings to the lake would probably result in the rapid improvement of the water quality of the Western basin, southshore region of the Central basin, and both the south and northshore areas of the Eastern basin; a much slower improvement would be expected for the offshore waters of the Central basin. The improvement of the offshore Eastern basin waters would be so slow as to be almost undetectable.

The Detroit River is the major input source of both phosphorus and chloride to Lake Erie. The chloride model demonstrates that the distribution of these materials in the Western basin is sensitive to the bank of the river from which the loading originates. It appears that the major chloride input is from the east bank while the major phosphorus input is from the west bank. Further, the chloride model results have verified the water transport model developed for the 1970 shipping season and have established the magnitude of the diffusion coefficient (for a specific time and space scale) for dissolved materials in Lake Erie.

Wind-induced wave turbulence is a process of major importance in Lake Erie because of the long fetch in the direction of the prevailing wind (450 km) and the shallow basin depths (8-, 18-, 27-m mean depths). The wave generating process has been modeled and combined with the advective–diffusive model to produce a phosphorus re-

generation model based on sedimentological data, while neglecting biological processes. This phosphorus model produces a good simulation of observed results of phosphorus concentration during a storm period in 1970 and stresses the importance of physical processes in phosphorus models. The biological processes would be of major importance under conditions of calm weather and thermal stratification.

The development of a complete phosphorus budget for Lake Erie has been attempted utilizing the results of the studies included here on the river, erosional, and sediment inputs and the forms of phosphorus found in the eroding materials and sediments. A reasonable agreement has been obtained between the estimate of net loadings of phosphorus to the lake and the estimated loading of phosphorus to the sediments. Precolonial (prior to 1850) loadings of phosphorus to the lake consisted of approximately 77% of the phosphorus being in the apatite form while the present phosphorus loadings to the lake consist of about 50% in the apatite form. Large quantities of phosphorus were found to have disappeared from the waters of the Western basin and subsequently to have accumulated in the sediments of the Eastern basin. Although a number of different transport mechanisms have been suggested, it has not been possible to define the process by which this phosphorus was transported through the lake.

The knowledge of biological processes within the lake is significantly less than the knowledge of nonbiological processes. However, the lake-wide, phytoplankton and zooplankton data included in this special issue are more extensive than other data on Lake Erie which have been published to date. These new reports have established the nature of the phytoplankton and zooplankton communities in the lake at the present time.

There is a definite decrease in average algal densities from west to east in the lake. The species composition of phytoplankton was constant throughout the year while the relative abundance of the different species changed markedly. Thermal stratification appears to have a major effect on the algal species abundance, causing the diatom dominance to be almost completely replaced by flagellates in the Eastern basin with a similar replacement of reduced magnitude in the Central basin during the summer. In the unstratified Western basin, diatoms remain the major species all year with an increase in the blue-green fraction towards the end of the summer. Phytoplankton size analysis indicated that in all three basins the nannoplankton dominated the phytoplankton community, and the size fractionation experiments confirmed their physiological importance as they were found to contribute overwhelmingly to the total productivity.

Crustacean zooplankton vary widely in species composition and abundance both horizontally and seasonally over the lake. West to east gradients in abundance appear to be related to the difference in the timing of peaks of abundance associated with the time lag in temperature change between basins. The dominant species of crustacean zooplankton in Lake Erie are characterized by short generation times and high reproductive potential. Changes in abundance of some species are largely controlled by the seasonal replacement of one species by another. However, the abundance of at least one of the major groups, the cladocerans, is clearly related to the presence of particulate organic matter and results in south to north concentration gradients in the Central basin.

The fish in Lake Erie have been affected by humans in a number of different ways. The increased cultural eutrophication has contributed to the disappearance of some species. Landfill and land-clearing activities, resulting in increased siltation, have also seriously damaged many of the spawning beds of different species of fish. However, the greatest human effect on Lake Erie fish has been overfishing.

LAM, D. C. L., AND J.-M. JAQUET 1976. Computations of physical transport and regeneration of phosphorus in Lake Erie, fall of 1970. J. Fish. Res. Board Canada, 33: 550–563.

Appendix 1.

REFERENCES RELATING TO BENTHIC FAUNA IN LAKE ERIE

BEETON, A. M. 1969. Changes in the environment and biota of the Great Lakes, p. 150–187. *In* Eutrophication: causes, consequences, correctives. Natl. Acad. Sci., Washington, D.C.

BRINKHURST, R. O. 1969. Changes in the benthos of Lakes Erie and Ontario. Bull. Buffalo Soc. Nat. Sci. 25: 45–65.

BRINKHURST, R. O., A. L. HAMILTON, AND H. B. HERRINGTON. 1968. Components of the bottom fauna of the St. Lawrence Great Lakes. Publ. Great Lakes Inst., Univ. Toronto Prog. Rep. 33: 49 p.

BRITT, N. W. 1955a. Stratification in western Lake Erie in summer of 1953: effects on *Hexagenia* (Ephemeroptera) population. Ecology 36: 239–244.

1955b. *Hexagenia* (Ephemeroptera) population recovery in western Lake Erie following the 1953 catastrophe. Ecology 36: 520–522.

CARR, JOHN F., AND J. K. HILTUNEN. 1965. Changes in the bottom fauna of Western Lake Erie from 1930 to 1961. Limnol. Oceanogr. 10: 551–569.

CHANDLER, DAVID C. 1963. Burrowing Mayfly nymphs in western Lake Erie prior to 1947. Great Lakes Res. Div., Univ. Mich. Publ. 10: 267–268.

WOOD, KENNETH G. 1963. The bottom fauna of western Lake Erie 1951–52. Great Lakes Res. Div., Univ. Mich. Publ. 10: 258–265.

COMING NEXT MONTH / À PARAÎTRE LE MOIS PROCHAIN

PART 1

Effects of low concentrations of oil accommodated in sea water on the growth of unialgal marine phytoplankton cultures **N. J. Prouse, D. C. Gordon Jr., and P. D. Keizer**

Changes in blood chemistry and blood cell morphology in coho salmon (*Oncorhynchus kisutch*) following exposure to sublethal levels of total residual chlorine in municipal wastewater **J. A. Buckley, C. M. Whitmore, and R. I. Matsuda**

Food and food selectivity of the black bullhead, *Ictalurus melas*, in Lake Poinsett, South Dakota **A. Repsys, R. L. Applegate, and D. C. Hales**

Distribution of sound scattering layers caused by euphausiids and their relationship to chlorophyll *a* Concentrations in the Gulf of St. Lawrence Estuary **D. D. Sameoto**

Year-class formation in the walleye (*Stizostedion vitreum vitreum*) population of Oneida Lake, New York, 1966–73 **J. L. Forney**

Optimal controls for a single species fishery and the economic value of research **C. C. Huang, I. B. Vertinsky, and N. J. Wilimovsky**

The number of genes for lactate dehydrogenase in salmonid fishes **G. S. Bailey, H. Tsuyuki, and A. C. Wilson**

Agonostic behavior of captive goldeye (*Hiodon alosoides*) **D. A. Fernet and R. J. F. Smith**

Seasonal mortality of rainbow trout (*Salmo gairdneri*) planted in small eutrophic lakes of Central Canada **G. B. Ayles, J. G. I. Lark, J. Barica, and H. Kling**

Thermal tolerance of stored *Fundulus heteroclitus* gametes: fertilizability and survival of embryos **K. P. Kuchnow and R. S. Foster IV**

Assessment of the United States Whitefish Fisheries of Lake Superior, Lake Michigan, and Lake Huron **A. L. Jensen**

Salinity effects on plasma electrolytes of channel catfish, *Ictalurus punctatus* **K. B. Davis and B. A. Simco**

Hepatic biotransformation of environmental xenobiotics in six strains of rainbow trout (*Salmo gairdneri*) **M. G. Pedersen, W. K. Hershberger, P. K. Zachariah, and M. R. Juchau**

Occurrence of R factors associated with *Aeromonas hydrophila* isolates from aquarium fish and waters **E. B. Shotts Jr., V. L. Vanderwork, and L. M. Campbell**

Aspects of the biology and morphology of the snake eel, *Pisodonophis cruentifer* (Pisces, Ophichthidae) **C. A. Wenner**

Electrophysiological studies of morpholine-imprinted coho salmon (*Oncorhynchus kisutch*) and rainbow trout (*Salmo gairdneri*). **J. C. Cooper and A. D. Hasler**

Bacterial flora of aquarium fishes imported from Southeast Asia **E. B. Shotts Jr., A. L. Kleckner, J. L. Blue, and J. B. Gratzek**

Infection of lumpfish (*Cyclopterus lumpus*) with larvae and of Atlantic cod (*Gadus morhua*) with adults of the copepod, *Lernaeocera branchialis*, in and adjacent to the Newfoundland area, and inferences therefrom on inshore–offshore migrations of cod **W. Templeman, V. M. Hodder, and A. M. Fleming**

Experimental confirmation of the olfactory hypothesis with homing artificially imprinted, coho salmon (*Oncorhynchus kisutch*) **J. C. Cooper, A. T. Scholz, R. M. Horrall, A. D. Hasler, and D. M. Madison**

Notes

Homing of artificially imprinted steelhead (rainbow) trout, *Salmo gairdneri* **J. C. Cooper and A. T. Scholz**

Developmental stages of *Pontella meadi* Wheeler (Copepoda: Calanoida) **V. R. Gibson and G. D. Grice**

Diel ecdysis rhythms in juvenile lobsters (*Homarus americanus*) **G. R. Tamm and J. S. Cobb**

Distribution and timing of tag recoveries from native and nonnative Atlantic salmon (*Salmo salar*) released into Big Salmon River, New Brunswick **B. M. Jessop**

Protection of steelhead trout (*Salmo gairdneri*) against vibriosis with a living low-virulence strain of *Vibrio anguillarum* **B. A. Braaten and H. O. Hodgins**

System for sampling large trawl catches of research vessels **S. E. Hughes**

An instrument with internal calibration for monitoring chlorine residuals in natural waters **G. Marienko, R. J. Huggett, and D. G. Friend**

Growth and survival of young-of-the-year emerald shiners (*Notropis atherinoides*) at different temperatures **J. H. McCormick and C. F. Kleiner**

Mixed species schooling of postlarvae of *Ammodytes hexapterus* and *Clupea harengus harengus* **S. W. Richards**

Perspectives

Structured aquaculture development with a Canadian perspective — G. I. Pritchard

Book Reviews/Comptes rendus

The pathology of fishes (W. E. Ribelin and G. Migaki [ed.]). — L. Leibovitz
The biology of the Indian Ocean (B. Zeitzeschel [ed.]). — J. C. Marr

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PART 2

Aquaculture Symposium — 13th Pacific Science Congress 1975

Preface/Préface — E. M. Donaldson

Economic and Legal Considerations in Aquaculture

Aquaculture revisited — J. E. Bardach
Engineering and the economics of aquaculture — P. H. Milne
Role of aquaculture in economic development within Southeast Asia — A. Sribhibhadh

Current Status and Future Prospects for Culture of Economically Important Species

Farming ocean ranges for salmon — T. Joyner
Systems of polyculture of fishes in the inland waters of India — V. G. Jhingran
Yellowtail (*Seriola quinqueradiata*) farming in Japan — M. Fujiya
Status and potential of pond culture of penaeid prawns in Southeast Asia — K. Kuronuma
Culture of milkfish (*Chanos chanos*) as a means of increasing animal protein supply — T. P. Chen

Endocrinology and Reproductive Physiology in Aquaculture

Hormones and the control of reproduction in aquaculture — M. Fonatine
Use of hormones in induced spawning of carps — H. Chaudhuri
Application for hormones in fish culture — F. Yamazaki
Photoperiod and temperature control in the regulation of reproduction in the female catfish *Heteropneustes fossilis* — B. I. Sundararaj and S. Vasal
In vitro oocyte maturation and ovulation in rainbow trout (*Salmo gairdneri*), northern pike (*Esox lucius*), and goldfish (*Carassius auratus*) — B. Jalabert
Endocrine control of ovulation in medaka (*Oryzias latipes*) and ayu (*Pecoglossus altivelis*) — K. Hirose
Cryopreservation of fish spermatozoa and ova — H. F. Horton and A. G. Ott

Current Status and Future Prospects for Seaweed Culture

Progress of *Laminaria* cultivation in Japan — Y. Hasegawa
Role of mariculture in the Canadian seaweed industry — I. C. Neish
Aquacultural techniques for creating and restoring beds of giant kelp, *Macrocystis* spp. — W. J. North
Current status and future prospects for harvesting and resource management of the agarophyte in Japan — N. Yamada

Health, Nutrition, and Growth in Aquaculture

Formulating practical diets for fish — J. E. Halver
Use of synthetic diets and biochemical criteria in the assessment of nutrient requirements of fish — C. B. Cowey
Factors affecting maximization of fish growth — A. H. Weatherley
Prevention and control of viral diseases of salmonids — D. F. Amend

Genetics in Aquaculture

Mariculture — ecological and genetic aspects of production — J. R. Calaprice
Genetic techniques in flatfish culture — C. E. Purdom
Possibilities for genetic improvements in salmonids — T. Gjedrem
Review of genetic and related studies on commercial oysters and other pelecypod molluscs — A. C. Longwell
Applications of biochemical genetics to aquaculture — D. Hedgecock, R. A. Shleser, and K. Nelson

Journal of the Fisheries Research Board of Canada

Guide for Authors

A more detailed guide on preparation of manuscripts from theses is available from the Office of the Editor.

Content

We invite submission of manuscripts in the broad field of fisheries management, ocean science, and aquatic environments relevant to Canada.

Articles: Reports of research studies of broad scope that are original contributions to science.

Notes: Differ from papers on the basis of scope or completeness, rather than quality of research. They may be on problems with narrow, readily defined limits; on work that is largely confirmatory; on methodology or research techniques; on advances in knowledge arising as by-products of broader studies; on progress in projects that were discontinued; or on preliminary findings that warrant publication before problems are resolved. They may be organized with as much flexibility as articles except that they have no centered headings and should occupy no more than four printed pages.

Reviews: Detailed articles that are critical appraisals of broad areas of investigation, particularly in fields that have shown rapid development in recent years.

Perspectives: Essays of opinion or hypothesis on fisheries, ocean science, aquatic environments, or related topics that are of concern to the professional and layman. References are desirable; informal style is welcomed.

Letters: Opinion on topics recently under discussion in the *Journal*; preferred maximum length, 500 words. The Editor reserves the right to shorten letters and to make other changes to accord with the style of the *Journal*.

Editorial Practices

The *Style Manual for Biological Journals* (3rd ed. American Institute of Biological Sciences, 3900 Wisconsin Ave. NW., Washington, D.C.) is used as the authority in matters of form. The standards for spelling are: *Webster's Third New International Dictionary* and *le Grand Larousse Encyclopédique*. Titles of periodicals are abbreviated as in the *Biological Abstracts* list. References to literature are made by the name-and-year system. The *Journal* follows the names and spelling recommended in *A List of Common and Scientific Names of Fishes from the United States and Canada* (3rd ed. 1970) Special Publication No. 6 of the American Fisheries Society.

Series titles should be avoided. An *abstract* is required for all manuscripts, and should state concisely, in less than 250 words, what was done, the essentials of the new knowledge, and the main conclusions. Abstracts are published in English and French; those submitted in one language only are translated by the *Journal*.

The Editor normally submits each manuscript to at least two *referees*. Contributors are encouraged to suggest at least four names to be added to the list from which referees may be selected.

Illustrations should be submitted as photographic reproductions no larger than a standard page and must accompany the manuscript. Prints should not be mounted, stapled, or clipped. Symbols, letters, and numerals must be neat, and at least 1.5 mm high when the illustration is reduced to the appropriate size. Drawings should, when possible, be grouped to facilitate comparisons.

Proofs are sent to the author, who is responsible for correcting only errors in typesetting. *Alterations to the content must be avoided.*

Fifty *reprints* are supplied free. Additional reprints may be purchased at the time of publication (an order form is sent out with the galley proof). All orders should be sent to the Printing Department, University of Toronto Press, 5201 Dufferin Street, Downsview, Ont. M3H 5T8. Invoices relating to the reprints will be sent out by the Printer. A charge of $40.00 per page is made for articles published. When submitting a manuscript, an author should state that his institution will accept this charge.

Checklist on Manuscripts

Attention to the following questions will expedite appraisal of manuscripts by referees and editors.

Are the findings, interpretations, and conclusions adequately documented and relevant to the purpose of the study?

Are all the tables and figures needed, and organized to facilitate comparisons? Are there inconsistencies between tables or figures and the text, or within the text? Should some of the data be made available separately, in a manuscript report or at a data repository? If you use statistical analysis are the statistics subordinate to the research?

Would any of the text be clearer if condensed? Are summary statements given at the beginning of sections and paragraphs, and are details in sections and paragraphs relevant to their topics? Does the organization of the manuscript follow logically from the statement of purpose in the introduction?

Does the *title* encompass the content of the report?

Does the *abstract* give the essentials of the new knowledge?

Is the *introduction* largely limited to the scope, purpose, and rationale of the study? Is review of the literature limited to defining the problem?

Are details of *materials and methods* limited to what scientists need in understanding the design of the study and in judging adequacy of the data?

Are generalizations on the *results* adequate? Are findings distinguished from inferences from them?

Is *discussion* limited to interpretation and significance of the findings?

Journal de l'office des recherches sur les pêcheries du Canada

Guide des Auteurs

On peut se procurer au bureau du Directeur de la publication un guide plus détaillé pour la préparation de manuscrits à partir de thèses.

Sujets

Nous désirons reçevoir des manuscrits s'insérant dans le vaste domaine de la gestion des pêches, des sciences de la mer et des milieux aquatiques au Canada.

Articles: Comptes rendus d'études d'une portée générale, constituant des contributions inédites à la science.

Notes: Se distinguent des articles par la portée et le degré d'achèvement plutôt que par la qualité des recherches. Elles peuvent traiter de problèmes particuliers et bien définis; de travaux qui sont en grande partie confirmatoires; de méthodologie ou de techniques de recherche; de progrès de la connaissance découlant d'études plus vastes; de développements dans des projets abandonnés; ou de découvertes préliminaires qui méritent d'être publiées avant la solution de problèmes à l'étude. Les notes peuvent être présentées d'une façon aussi souple que les articles, mais elles ne doivent pas porter d'en-têtes centrés ni occuper plus de quatre pages.

Critiques: Articles détaillés constituant une appréciation critique de vastes champs de recherches, particulièrement dans les domaines ayant connu de rapides progrès au cours des dernières années.

Perspectives: Essais de l'opinion ou de l'hypothèse sur les pêches, les sciences de la mer, les milieux aquatiques, ou sur des sujets connexes qui intéressent professionnels et profanes. Il est préférable de donner des références et d'écrire, si possible, dans un style simple.

Lettres: Opinions sur des sujets exposés dans le *Journal*. Longueur maximale préférable: 500 mots. Le directeur se réserve le droit de raccourcir les lettres et de faire d'autres modifications dans le style du *Journal*.

Dispositions générales

Le *Style Manual for Biological Journals* (3e éd., publié par l'American Institute of Biological Sciences, 3900 avenue Wisconsin N.O., Washington, D.C.) sert de guide quant à la forme de présentation. Le *Webster's Third New International Dictionary* et le *Grand Larousse Encyclopédique* font autorité pour l'orthographe des mots. Les titres de périodiques sont abrégés suivant les *Biological Abstracts*. Les références bibliographiques dans le texte doivent suivre le système nom de l'auteur-année. Pour les noms de poissons et leur orthographe, le *Journal* suit les recommandations contenues dans *A List of Common and Scientific Names of Fishes from the United States and Canada* (Troisième édition, 1970), Publication spéciale No. 6 de l'American Fisheries Society.

Il faut éviter les *articles en série*. Un *résumé* doit accompagner chaque texte et doit exposer brièvement (moins de 250 mots) le travail accompli, les points essentiels des connaissances acquises et les principales conclusions. Les résumés paraissent en anglais et en français, le *Journal* veillant à la traduction dans le cas des présentations unilingues.

Le directeur soumet d'ordinaire chaque texte à l'approbation de deux *arbitres* ou plus. Nous encourageons les collaborateurs à suggérer au moins quatre personnes dont les noms pourraient figurer sur la liste des arbitres.

Les *illustrations* doivent être sous forme de reproductions photographiques ne pas dépasser les dimensions d'une page normale et accompagner le texte. Elles ne doivent pas être montées, coupées ni brochées. Les symboles, lettres ou chiffres doivent se détacher avec précision et avoir au moins 1,5 mm de hauter lorsque l'illustration est réduite à la grandeur voulue. Les dessins doivent, autant que possible, être groupés pour faciliter l'étude comparative.

L'auteur doit revoir les *épreuves* qu'on lui remet et en corriger seulement les erreurs typographiques. *Il doit éviter de changer le contenu du texte.*

Cinquante *tirés à part* sont fournis gratuitement. Les tirés à part peuvent être achetés au moment de la publication. Un bon de commande à cet effet accompagne les épreuves; il doit être retourné en même temps que celles-ci, non pas au bureau de la rédaction, mais au Printing Department, University of Toronto Press, 5201 Dufferin Street, Downsview, Ont. M3H 5T8, qui se chargera de la facturer. Les frais sont de $40.00 par page pour la publication des articles. Quand un auteur soumet un manuscrit, il doit spécifier que son institution accepte les frais.

Listes de contrôles pour manuscrits

Une attention particulière aux questions suivantes facilitera l'évaluation des manuscrits par les arbitres et les éditeurs.

Les découvertes, interprétations et conclusions sont-elles bien documentées et pertinentes au but de l'étude?

Les tableaux et figures sont-ils tous nécessaires et agencés pour faciliter les comparaisons? Y a-t-il contradiction entre les tableaux ou les figures et le texte, ou dans le texte même? Est-ce que certaines données devraient être présentées séparément, dans un rapport manuscrit ou dans un dépôt de données? Si vous faites une analyse statistique, la statistique cède-t-elle le pas à la biologie?

Le texte y gagnerait-il en clarté s'il était plus condensé, en tout ou en partie? Contient-il des exposés sommaires au début de chaque section ou paragraphe, et les détails de ces sections et paragraphs son-ils pertinents au sujet traité? La marche du manuscrit est-elle conforme à la logique énoncée dans l'introduction?

Le *titre* est-il concis, tout en étant représentatif du contenu du rapport?

Le *résumé* donne-t-il l'essentiel des connaissances nouvelles?

L'*introduction* se limite-t-elle à une esquisse de la portée, du but et des raisons de l'étude? La revue de la littérature se limite-t-elle à la définition du problème?

La section *matériel et méthode* contient-elle le minimum de détails qui permettront aux autres scientifiques de saisir les grandes lignes de l'étude et de juger de la valeur des données?

Les généralisations sur les *résultats* sont-elles adéquates? Permettent-elles de distinguer entre constatations et déductions?

La *discussion* est-elle limitée à l'interprétation et à l'évaluation des découvertes?

Journal of the Fisheries Research Board of Canada

Guide for Typists Preparing Copy

Read "Guide for Authors" in the current volume of the *Journal*. When in doubt about how to type or arrange details, follow *Journal* practice.

Type the manuscript on white bond paper, not flimsy, preferably size 21.5 × 28 cm (8.5 × 11 inches). If possible, use paper with the lines numbered in the margin. Leave all margins at least 2.5 cm wide.

Double-space all parts of the manuscript, including tables, captions for illustrations, footnotes, and the reference list. Underline only material that is to be printed in italics. Indent the first line of each paragraph in the text and of each footnote.

In titles and headings, capitalize the first letter of each word. For small capitals, use small letters.

Indicate all text footnotes by superscript arabic numerals. Number them consecutively throughout the manuscript, starting with those on the title page. Place each footnote immediately below the reference to it, separating it from the text by horizontal lines.

On the first page type only the title, the authors' names and affiliations, related footnotes, and the running head (see Appendix A). Type the abstract on a separate page (see Appendix B). Number all pages, beginning with the title page and including those for tables and captions for illustrations.

Have the author check the finished manuscript, and send the ribbon copy and two other clear copies to the Office of the Editor.

Tables: Examine the current volume of the *Journal* to see how tables are set up (see Appendix C).
Note especially the following concerning *numbered tables:*

Type each on a separate page, using paper without the lines numbered.

Number them with arabic numerals.

Use horizontal lines above and below the headings and below the columns, and seldom elsewhere. Never use vertical lines; leave extra spaces instead.

Indicate table footnotes by superscript small letters and type them below the table.

Place the tables after the list of references.

Where each table is first referred to, type in the margin "Table — near here."

Note that *text tables* are not numbered, are typed with the text, and seldom need horizontal lines (see Appendix C).

Illustrations: Send a photographic reproduction (no larger than 21.5 × 28 cm) of each illustration with the manuscript. Type the author's name and figure number on a strip of paper and fasten it to the back of the print. Do not write or type on the print itself. Also, type the captions in succession on a separate page, headed "Captions for illustrations." Put this page at the end of the manuscript.

Where each figure is first referred to, type in the margin "Fig. — near here."

Mathematical Expressions, Equations, and Formulas: Leave three spaces above and below lines that have much material in superscripts, and above and below equations and formulas. In the various expressions, place each component in its correct position (see Appendix D).

Show all symbols clearly. In the margin, spell out Greek letters the first time they occur, and distinguish between the small letter *l* and the numeral *one*, and between the capital letter *O* and the numeral *zero*. Make other explanatory notes similarly.

References: Check the text to see that all references in the list are referred to. Type the reference list, double-spaced, in the style used in the *Journal* (see Appendix E). Names of journals are abbreviated as in the *Biological Abstracts* list; if an abbreviation is in doubt, type the name in full.

Spellings: About 100 words common in fisheries journals are listed in Appendix F.

Appendices: Appendices A to F are available on request from the Office of the Editor.

Journal de l'office des recherches sur les pêcheries du Canada

Guide pour les secrétaires d'auteurs

Lisez le "Guide des Auteurs" reproduit dans l'édition courante du *Journal*. Si vous avez des doutes sur la façon de dactylographier le texte ou d'en agencer les détails, suivez la pratique du *Journal*.

Dactylographiez le manuscrit sur papier blanc de bonne qualité, et non sur papier pelure, mesurant de préférence 21,5 × 28 cm (8,5 × 11 pouces), laissant toujours une marge d'au moins 2,5 cm. Si possible, employez un papier avec lignes numérotées en marge.

Tout le manuscrit, y compris tableaux, légendes de figures, renvois et références bibliographiques, devra être dactylographié à double interligne. Ne soulignez que les mots qui devront être imprimés en italique. Mettez en retrait la première ligne de chaque paragraphe du texte et de chaque renvoi.

Dans les titres et les en-têtes, n'employez la majuscule que pour le premier mot et les noms propres. Pour les petites majuscules, employez les miniscules du dactylographe.

Indiquez tous les renvois du texte par des chiffres arabes suscrits. Numérotez les renvois consécutivement par tout le texte, à commencer par ceux de la page titre. Placez chaque renvoi directement sous sa référence, le séparant du texte par des lignes horizontales.

La première page ne doit contenir que le titre de la publication, ne nom et l'affiliation des auteurs, les renvois appropriés et le titre courant (voir Appendice A). Dactylographiez le résumé sur une page séparée (voir Appendice B). Numérotez chaque page, à commencer par la page titre, et y compris celles qui contiennent les tableaux et la légende des figures.

Voyez à ce que l'auteur vérifie le manuscrit final, et soumettez la première copie et ainsi que deux copies lisibles au bureau du Directeur de la publication.

Tableaux. Consultez une copie du *Journal* pour la composition des tableaux (voir Appendice C).

Portez une attention particulière à ce qui suit au sujet des *tableaux numérotés*.

Dactylographiez chaque tableau sur feuille séparée, vous servant de papier sans lignes numérotées.

Numérotez les tableaux en chiffres arabes.

Tracez des lignes horizontales au-dessus et au-dessous des en-têtes et au-dessous des colonnes, rarement ailleurs; ne tracez jamais de lignes verticales; laissez plutôt des espaces supplémentaires.

Indiquez les renvois dans les tableaux par les minuscules suscrites et placez les renvois au bas du tableau.

Placez les tableaux à la suite des références bibliographiques.

A l'endroit où le tableau est mentionné pour la première fois, indiquez dans la marge "Tableau — près d'ici."

Remarquez que les *tableaux dans le texte* ne sont pas numérotés, sont dactylographiés avec le texte et ne requièrent que rarement des lignes horizontales (voir Appendice C).

Figures: Envoyez avec le manuscrit un reproduction photographique (21,5 × 28 cm ou moins) pour chaque illustration. Dactylographiez le nom de l'auteur et le numéro de la figure sur une lisière de papier que vous collerez au dos. N'écrivez pas à la main ou à la machine sur la photographie même. En plus, dactylographiez les légendes, dans l'ordre, sur feuille séparée, intitulée "Légendes des figures." Intercalez cette feuille à la suite du texte.

A l'endroit où chaque figure est mentionnée pour la première fois, inscrivez en marge "Fig. — près d'ici."

Equations, formules et expressions mathématiques: Laissez trois espaces au-dessus et au-dessous des lignes renfermant des chiffres ou des lettres suscrits et souscrits, ou encore au-dessus et au-dessous d'équations et de formules. Placez chaque composante des différentes expressions dans la position qui lui est propre (voir Appendice D).

Indiquez clairement tous les symboles. Epelez en marge les lettres grecques, la première fois qu'elles apparaissent. Faites une distinction entre la lettre *l* et le chiffre *un*, et entre la majuscule *O* et le chiffre *zéro*. Insérez de la même façon toute autre note explicative.

Références: Vérifiez le texte afin de vous assurer que toutes les références apparaissant dans la liste bibliographique sont bien mentionnées dans le texte. Dactylographiez à double interligne dans le style en usage dans le *Journal* (voir Appendice E). Les noms de revues sont abrégés suivants les *Biological Abstracts*; en cas de doute, écrivez le nom au long.

Orthographe: Une liste d'environ 100 mots communément employés dans les périodiques sur les pêches est donnée en Appendice F.

Appendices: On peut se procurer une copie des Appendices A à F sur demande au bureau du Directeur de la publication.

Journal of the Fisheries Research Board of Canada

Journal de l'Office des recherches sur les pêcheries du Canada

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